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**MORTALITY OF SEA TURTLES IN THE HAWAII LONGLINE FISHERY:
A PRELIMINARY ASSESSMENT OF POPULATION IMPACTS**

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SCOPE OF THE DOCUMENT

National Marine Fisheries Service (NMFS) is conducting an internal consultation on the Hawaii longline fishery under Section 7 of the Endangered Species Act. The Consultation will produce a (revised) Biological Opinion on the fishery specifying acceptable levels of incidental mortality to turtles and other protected species that may be caused by Hawaii longline fishing operations. The purpose of this document is to provide certain background information and analyses in support of the Consultation. In particular, it summarizes results of some provisional studies using a turtle population simulation model, TURTSIM (Wetherall, 1996a), to assess impacts of the longline fishery on sea turtle populations. It also discusses procedures for establishing annual maximum allowable incidental mortality levels consistent with population recovery goals and allowable mortality guidelines for the Hawaii longline fishery.

The results extend and update earlier findings using the TURTSIM model (Wetherall, 1996b) at an international workshop convened by NMFS in Honolulu during 4-7 December 1995 (Bolten et al., 1996; the "Workshop"). The Workshop dealt primarily with the status of Japanese loggerheads, which constitute most of the incidental take. Some attention was also given to leatherbacks, as comprehensive nesting data were presented for the endangered Malaysian leatherback population.

Since the Workshop, information has been published from surveys of Pacific leatherback nesting in Mexico indicating a severe decline in the number of nesters there. Accordingly, special attention is given in this report to assessing impacts of the Hawaii fishery on Mexican leatherbacks. Updated results are provided for Malaysian leatherbacks and Japanese loggerheads, taking into account new information and modifications to the TURTSIM code.

The analyses described here, like those in the Workshop document, are highly speculative. Most of the life history parameters and other model input parameters are still little more than guesses and critical assumptions are untested. Accordingly, caution is advised in interpreting and applying the results.

POPULATIONS ADDRESSED AND STOCK STRUCTURE

Five species of sea turtles inhabit the region of the Pacific fished by Hawaii-based longline vessels:

Loggerhead	<i>Caretta caretta</i>
Leatherback	<i>Dermochelys coriacea</i>
Olive ridley	<i>Lepidochelys olivacea</i>
Green turtle	<i>Chelonia mydas</i>
Hawksbill	<i>Eretmochelys imbricata</i>

To monitor interactions with turtles in the Hawaii longline fleet, a NMFS observer program has been in place since 24 February 1994. Observers monitored 1,700 fishing operations (sets) involving 1.9 million hooks through December 1996, recording a total of 106 interactions with sea turtles. The species composition of the observed turtle takes, based on aggregate data, was as follows:

<u>Species</u>	<u>Observed Interactions</u>	
	<u>Number</u>	<u>Percent</u>
Loggerhead	56	52.8
Leatherback	21	19.8
Olive ridley	16	15.1
Green turtle	5	4.7
Hawksbill	0	0.0
Unidentified hardshell	8	7.5
Total	106	99.9

Knowledge of stock structure and natal origins of turtles encountered by the fishery are critical to the assessment of fishery impacts. Information on stock structure is just now emerging. The current information with respect to the central North Pacific may be summarized as follows:

1. Loggerheads - Analysis of MtDNA from loggerheads sampled by high-seas driftnet fishery observers in the North Pacific during 1990-91 suggests that the loggerheads taken in the Hawaii-based longline fishery almost surely originate from nesting beaches in Japan (Bowen et al., 1995); 93% of the samples analyzed were from Japan and 7% from the Australian loggerhead population.
2. Leatherbacks - MtDNA evidence indicates that leatherbacks taken in the Hawaii longline fishery originate from multiple sources. Of the four specimens examined so far, two were identified with the western/Indo-Pacific (at least one with the Irian Jaya haplotype), and the other two were linked to Eastern Pacific rookeries (Mexico or Costa Rica, probably the latter) (Dutton et al., 1997; Dutton, pers. commun.). Preliminary analyses of allele frequency data from the four principal nesting beaches in Mexico suggest a single, random-mating population (Barrangan et al., 1997). Further, there appears to be a high degree of gene flow between rookeries in Mexico and Costa Rica.
3. Green turtles - Likewise, green turtles taken in the Hawaii fishery appear to originate from multiple sources; two of three specimens from Hawaii and one from the Eastern Pacific (Dutton et al., 1997).

4. Olive ridleys - Genetic material from olive ridleys taken in the Hawaii longline fishery has also been analyzed but interpretation of results is complicated by a lack of distinctive geographic structuring in the eastern Pacific olive ridley rookeries (Balazs, pers. commun.).
5. Hawksbills - No hawksbills have been identified so far in observed Hawaii longline operations. Given the depauperate state of the Hawaii hawksbill population (the U.S. Fish and Wildlife Service (USFWS) records about 15 nesters per year) and most other nesting populations in the equatorial Pacific (Eckert, 1993) an encounter with this species in the offshore longline fishing grounds would be rare. During 2 years of international observer effort on North Pacific high-seas driftnet vessels, only a single hawksbill take was documented.

The TURTSIM analyses in this document and the Workshop report focus on loggerheads and leatherbacks, which together make up at least 73% of the turtle takes. Brief attention is also given to Hawaiian green turtles, for which biological data and historical nesting data are readily available. Analyses of other green turtle populations (e.g., Mexico, Japan) that may interact with the Hawaii longline fishery, olive ridleys, and hawksbills have not yet been done. Although many olive ridley populations have been greatly reduced from pristine levels, recent surveys indicate that some populations are recovering after harvesting was reduced or banned. Fairly extensive nesting is now being reported at some rookeries. For example, about 700,000 nests (roughly 350,000 nesters) were reported at La Escobilla beach, Oaxaca, Mexico, each year during the 1994-95 and 1995-96 nesting seasons (Márquez-M., 1996), a significant increase over nesting levels during the previous 2 decades. Similarly, during the 1994 nesting season more than 80,000 olive ridleys participated in arribadas at Playa Nancite, Costa Rica (Gates et al., 1996). Nesting at the other major Costa Rican rookery, Playa Ostional, is typically an order of magnitude greater. The magnitude of olive ridley mortality in the Hawaii longline fishery (Table 5) is probably negligible compared with other sources; nevertheless, the impact should be assessed.

A higher priority should be given to assessing impacts on hawksbills. The fact that no hawksbills have been recorded yet in catches monitored by NMFS observers suggests that the Hawaii longline fishery has little impact on this species. Considering that fewer than 5% of longline trips are monitored, however, a total fleet take of several hawksbills annually could easily go undetected. Moreover, relatively low levels of mortality could significantly affect the small Hawaii population.

LIFE HISTORY ASSUMPTIONS

In most areas of the Pacific, as in other regions of the world, little is known about the life history of sea turtles apart

from their nesting biology. Except were noted below, in the analyses reported here assumptions about the life history of Pacific leatherbacks and Japanese loggerheads were based on consensus findings of the Workshop (Bolten et al., 1996). These life history assumptions and parameters are summarized, with amendments, in Tables 1-4. With respect to the stage-specific survival rates, the values given in the tables are simply educated guesses based on consideration of the few cases worldwide where survival rates have been estimated from field studies; survival analyses have not been done for the populations of interest. Information is particularly scanty for leatherbacks. Data for Japanese loggerheads are somewhat more complete, with estimates available for parameters of growth, maturation, and reproduction. In general, remigration interval distributions used in TURTSIM were inferred from the estimated average percentages of adult females assumed to be nesting (given at the Workshop), conditional on the assumed levels of adult mortality.

In the present analyses, annual natural survival for adults was fixed at 95% and the early pelagic (denoted the "P1" stage) survival rate was estimated during the initialization step of the TURTSIM runs (see Wetherall, 1996a for an overview of TURTSIM procedures). Survival rates for the intermediate stages varied (Tables 1-4), typically adhering to values assumed at the Workshop. The survival assumptions appear to be reasonable, depicting a progressive reduction in natural mortality rate with advancing developmental stage; nevertheless, they are just educated guesses.

Where available, new information on biological parameters was used. In the case of leatherbacks, for example, the assumed age at maturity was reduced from 25 years to 14 years based on recent analyses of sclerotic ossicles by Zug and Parham (1996).

A significant modification of TURTSIM was made to allow for assumption of density-dependence in stage-specific growth rates, remigration probabilities, and other parameters. In the Workshop TURTSIM analyses, the only process allowing density dependence was the survivorship of eggs and hatchlings on the nesting beach; this was implemented by a Ricker model of hatchling production. In some circumstances (e.g., the highly publicized olive ridley *arribadas*), high density nesting occurs that may significantly reduce average egg survival. In most other situations, however, the effect of nester density on egg survivorship is probably relatively minor, and population regulation may be effected more by density dependence in other population processes. I am not aware of any case where density dependence has been demonstrated in sea turtles, except on a nesting beach. Low levels of density dependence would be almost impossible to detect, yet could exert significant influence on population dynamics.

In the present TURTSIM analyses, the following conditions were assumed with respect to density dependence:

- (1) Growth rates (annual growth transition probabilities) were inversely related to relative abundance within developmental

stages. Specifically, growth rates were assumed to be at a minimum when relative abundance within a stage was maximum and to increase smoothly to a higher rate as abundance was reduced. Let G denote the growth rate at population density D ; G_{\min} the minimum growth rate assumed to be achieved at a maximum population density, D_{\max} ; and G_{\max} the maximum growth rate achieved at low population densities. Specifically, in TURTSIM the relative growth rate was modeled as:

$$G/G_{\min} = 1 + \delta \{1 - (D/D_{\max})^{\beta}\}$$

where $\delta = (G_{\max} - G_{\min})/G_{\min}$ and β is the exponent controlling the shape of the relationship. In general, β was set equal to 10, so that the transition to slower growth rates occurred above about 75% of maximum population density (Fig. 1). This type of response has been demonstrated for marine mammals and large terrestrial mammals (e.g., Fowler, 1981; Fowler, 1990). In some comparisons a linear density dependence was assumed ($\beta = 1$). Growth probabilities under current low-density conditions were assumed to be 10% higher than under maximum densities (i.e., $\delta = 0.10$).

- (2) Remigration probabilities were inversely related to the relative abundance of adult females (with the same form as in growth).
- (3) Hatchling production was assumed to obey a Ricker function, but with a very weak density-dependent component; over the range of nester densities modeled, hatchling production was essentially linearly related to egg deposition.

HUMAN-INDUCED MORTALITY

To assess impacts of the Hawaii longline fishery it is necessary to account for mortality from all human activities. TURTSIM accommodates as many sources of human-induced mortality as desired and allows mortality levels to be specified by year, sex, and carapace length. TURTSIM also allows for separate human mortality to eggs and nesting females. Mortality of any type can be specified in terms of the absolute number of turtles killed, an exponential coefficient of mortality, or a scalar multiplier of the current mortality rate. The various options can be mixed to mimic very complex mortality scenarios.

In general, the TURTSIM simulations considered the following sources of human-induced mortality:

- Egg harvest
- Nester harvest
- Incidental fishing mortality
 - High-seas driftnet fisheries
 - Hawaii longline fishery
 - Other fisheries

Mortality arising from other human impacts, e.g., habitat loss or environmental degradation was not explicitly considered. The "other fisheries" category includes the distant water longline fisheries of Japan, Republic of Korea, Republic of China, Taiwan and other countries; tuna purse seine fisheries; and various coastal fisheries. Generally speaking, for most of the "other fisheries" reliable estimates of turtle take and mortality, incidental or otherwise, are unavailable (or data have not been published). Likewise, egg and nester harvests are rarely documented, particularly as they are now illegal in many countries. For a few fisheries, estimates of incidental take and mortality have been computed using a combination of fleet catch and effort statistics (e.g., from logbooks) and data collected through scientific observer programs (Table 5). Because observer programs usually monitor only a small fraction of the total fishing activity, however, estimates of turtle takes are often unreliable. Another problem in accessing fishery impacts is uncertainty about the mortality of turtles taken in the fishing gear, then released alive. Delayed effects of entanglement or hooking are generally unknown.

In the Pacific, sea turtles are subject to incidental capture in fishing gear deployed in coastal waters and on the high-seas. Coastal fisheries interacting with turtles deploy trawls, gill nets, and pelagic longlines. Turtle interactions with coastal fisheries generally are not monitored. In the U.S., however, NMFS observers have monitored interactions of marine mammals, seabirds and sea turtles in the California drift gillnet fishery and the California set gillnet fishery since July 1990. The drift gillnet fishery targets swordfish, thresher shark and mako shark, while the set gillnet fishery is directed towards Pacific angel shark and California halibut. Observer monitoring of the set gillnet fishery terminated after 1994 (although the fishery has continued). In both fisheries observer data and fleet effort statistics have been used to estimate total levels of turtle take and mortality (Julian and Beeson, in press). During 1990-95, point estimates of the annual take of leatherbacks in the drift gillnet fishery ranged from 6 to 32 turtles, including up to 26 per year (65% on average) that were dead upon retrieval of the gear (Table 5; Julian and Beeson, in press). Loggerheads were taken less frequently. In the set gillnet fishery, estimated takes and mortalities of turtles were much lower. Estimates for both fisheries are very imprecise because of the rarity of turtle interactions and relatively low observer coverage (12.4% of the drift gillnet operations through 1995 and 11.0% of set gillnet operations through 1994).

High-seas fishing gear taking turtles incidentally includes tuna purse seines and longlines set to catch tuna and billfish. High-seas drift gillnets took sea turtles but have not been deployed since December 1992, under a UN-sponsored moratorium. The high-seas fleets in the Pacific are multinational and, taken together, cover almost the entire pelagic range of sea turtles between the Subarctic Boundary in the North Pacific and the Subtropical Frontal Zone in the South Pacific. Although programs to document fishing effort and fish catch are in place for most of

these fleets, few programs provide for rigorous, systematic monitoring of sea turtle takes or interactions with other protected species. In particular, among the Pacific high-seas longline fisheries only the U.S. fishery based in Hawaii is closely monitored for turtle takes. To the best of our knowledge, similar programs have not been implemented in the much larger longline fleets of Japan, Taiwan, China, and South Korea.

In the North Pacific high-seas driftnet fisheries, observer data and fleet effort statistics indicate that 4,373 turtles, mostly loggerheads and leatherbacks, were entangled by combined fleets of Japan, Republic of Korea, and Taiwan during a 1-year interval from June 1990 through May 1991 when all fleets were monitored. Estimated mortalities were 1,011 turtles during the single year of comprehensive monitoring (Table 5).

In the Hawaii-based longline fishery, point estimates of total annual turtle take ranged from 441 to 688 during 1994-96, mostly loggerheads and olive ridleys (Table 5). Take estimates varied depending on year and estimation method. Estimates of annual turtle mortalities caused by the fishery have ranged from 67 to 106 (Skillman and Kleiber, in prep.). Although the take estimates vary between years and estimation methods, confidence ranges for total take are quite broad and overlap considerably because of the low precision involved; only about 5% of observer trips have been monitored (Skillman and Kleiber, in prep.).

In the eastern tropical Pacific (ETP), the Inter-American Tropical Tuna Commission (IATTC) has monitored incidental takes of sea turtles, along with other bycatch, in the purse seine fisheries that target primarily yellowfin tuna. Purse seine interactions with turtles are more likely during sets associated with floating objects (e.g., FADs, logs or other debris) that provide shelter and foraging opportunities for turtles than in sets associated with dolphins or free-swimming tuna (Hall, 1996). Based on observer data and logbook statistics, the IATTC estimates that the ETP purse seine fleet took a total of 295 turtles in 1993, and 167 turtles in 1994, mostly olive ridleys (Table 5; IATTC, 1997). An increased potential for bycatch of sea turtles was a side-effect of policies to reduce dolphin mortality by minimizing sets on tuna associated with dolphins.

In the TURTSIM analyses, various assumptions were made about historical and future human-induced mortality. When projections of dynamic behavior were generated (e.g., some of the leatherback analyses), estimates of mortality levels in the Hawaii longline fishery and North Pacific high-seas driftnet fisheries were used explicitly. Fishing mortality from other sources was lumped and simply adjusted arbitrarily to cause the TURTSIM projections to fit observed nester abundance data as closely as possible. Where mortality of eggs or nesters was involved, as in the leatherback analyses, available estimates were used (Malaysian leatherbacks) or arbitrarily adjusted to get the model to fit nester abundance (Mexican leatherbacks).

SIZE COMPOSITION OF FISHERY TAKES

A critical factor in assessing the impacts of fisheries is the size (or age) of turtles taken in the gear relative to the size at maturity in the population. Mortality of a subadult or adult turtle, particularly a nesting female, has a much greater impact on the population than death of an immature turtle (Crouse et al., 1987). Data on size composition of fishery takes have been collected in the various observer programs.

In the Hawaii longline fishery, observer measurements indicate that loggerheads, olive ridleys, and green turtles taken in the longline gear are immature (Fig. 2). Most leatherbacks taken also appear to be immature, but only the smallest of the 10 leatherbacks observed so far has been accurately measured; most leatherbacks are too large to be brought aboard safely. Maturity of most leatherbacks is uncertain as their carapace lengths were estimated by eye.

In the now defunct North Pacific high-seas driftnet fisheries, green turtles and the majority of loggerheads measured by observers were immature (Fig. 3). Of the leatherbacks observed in the monitored driftnet operations, those actually measured were immature. Some leatherbacks too large to be retrieved may have been mature, but estimates of their carapace lengths are unreliable. The single hawksbill observed in the high-seas driftnet fishery was an immature turtle with a 46 cm curved carapace length.

ANALYSES

The analyses described here augment or modify results reported at the Workshop (Wetherall, 1996b); the reader should refer to the Workshop report for background information. As in the Workshop analyses, two types of TURTSIM simulations were undertaken to assess impacts of humans on the turtle populations. In one type of analysis, TURTSIM was used to compute equilibrium characteristics of the turtle population under various sets of biological assumptions and hypothetical incidental take policies. This resulted in a set of steady-state relationships between key variables, e.g., between the coefficient of incidental mortality caused by pelagic fisheries and the equilibrium abundance of nesters or other population components. The equilibrium analyses may be used to estimate maximum allowable levels of human-induced mortality.

Another type of analysis involved simulating the annual dynamics of the population over time under a given set of assumptions with respect to life history characteristics and human activities. The results included historical and projected time trajectories of nester abundance and other measures of population viability in response to specified scenarios of incidental fishing

mortality and other human impacts. These simulations are useful for indicating the expected responses of the population to management interventions and illustrating the relative impacts of various components of human-induced mortality.

The TURTSIM models used in this assessment were largely deterministic and simulations were based on the best available estimates of population parameters and human-induced mortality. Studies that incorporate parameter uncertainty and demographic and environmental stochasticity are planned. These more rigorous TURTSIM simulations will assess recovery management strategies in a broader context and identify the levels of biological risk associated with incidental take decisions.

Japanese Loggerheads

Under the new assumptions about density dependence, the nominal growth and remigration parameters stipulated at the Workshop were interpreted to be those associated with current (i.e., low density) conditions. Current growth and remigration probabilities were assumed to be 10% higher than their minimum values under pristine (maximum density) conditions ($\delta = 0.10$). The density dependence parameter β was set to 10, and TURTSIM was initialized at the minimum rates (Table 1). The annual natural survival rate for P1 stage loggerheads was estimated at 78.8% by TURTSIM. An equilibrium nesting population of 10,000 loggerheads was assumed in the absence of human impacts.

Under the new assumption about adult survival (95% instead of 99% at the Workshop), the maximum sustainable P2 fishing mortality coefficient is about 0.046, less than estimated under the old loggerhead model, and the maximum sustainable fishing mortality is greater, about 7,000 P2 loggerheads (Fig. 4). The shape of the curve is also radically different than shown in the Workshop report due to the assumptions about density dependence. Theoretically, an equilibrium nester abundance equal to 75% of the maximum equilibrium nester abundance could be achieved if the annual P2 fishing mortality coefficient did not exceed about 0.035. Sustained incidental fishing mortality at higher levels would lead to a severe decline in equilibrium nester abundance.

Malaysian Leatherbacks

The model of leatherback nester dynamics at Terengganu, Malaysia, was recomputed under the revised assumptions on density dependence. As in the loggerhead case, estimates of remigration parameters stipulated at the Workshop were taken to reflect recent (low density) conditions. Likewise, nominal growth rates recently estimated by Zug and Parham (1996) from specimens taken in the eastern South Pacific were assumed to reflect current growth rates in the Malaysian population. Lower values of both parameters applicable to the virgin population were derived from the density-

dependence relationship and used to initialize the revised model (Table 2).

The revised trajectory of nester abundance shows a brief, transient increase of about 15-20 years after initialization of the simulated population as stage-specific densities are reduced by intensive harvesting of eggs (Chan and Liew, 1996; Wetherall, 1996b). This is followed by a collapse of nester abundance as the policy of harvesting eggs continues and other human impacts are added. As in the earlier version of the model, the policy of protecting some eggs and rearing them to hatchling stage was also simulated. Trajectories of the nesting population beyond 1997 were computed under three hypothetical scenarios concerning the Hawaii longline fishery. In each scenario, it was assumed that all other human-induced mortality was eliminated from 1998 onwards and a balanced hatchling sex ratio was restored (see Wetherall, 1996b):

Scenario 1 - All incidental mortality from Hawaii longline fishing was eliminated after 1997 so that only natural mortality remained.

Scenario 2 - The Hawaii longline fishery continued to operate at its current level of intensity, exerting the same coefficient of incidental fishing mortality on Malaysian leatherbacks as in 1997.

Scenario 3 - The coefficient of incidental fishing mortality on Malaysian leatherbacks generated by the Hawaii longline fishery expanded 5-fold over the current level (due to sudden expansion of fishing effort, increased vulnerability of leatherbacks, or other factors).

In the Scenario 2 simulations (status quo scenario), the current coefficient of incidental fishing mortality in the Hawaii longline fishery was set at a value equivalent to the upper 95% confidence limit on total leatherback mortality in the fishery, as estimated from observer and logbook data (Skillman and Kleiber, in prep.). It was assumed conservatively that all leatherback mortality caused by the Hawaii fishery involved turtles born in the Malaysian rookeries. Projected population responses suggest a very slow recovery of the population under the best conditions, and a potentially large impact of Hawaii longline fishing if the fishery affected Malaysian leatherbacks exclusively. Genetic data so far appear to rule out this possibility.

An equilibrium model for Malaysian leatherbacks was also computed. The results were similar in character to those for Japanese loggerheads. The P1 stage annual natural survival rate (in the absence of human impacts) was estimated by TURTSIM at 39.2%. The maximum sustainable coefficient of incidental fishing mortality in the P2 and adult stages is about 0.03 on an annual basis (Fig. 5). The maximum sustainable level of incidental fishing mortality in these life stages was judged to be about 450 leatherbacks per year, corresponding to a mortality coefficient of 0.024. The coefficient of incidental fishing mortality associated with a 75% level of relative nester abundance is about 0.018.

Mexican Leatherbacks

Trends in the Abundance of Nesters

During the 1995-96 Mexican leatherback nesting season, a comprehensive survey combining aerial and ground observations was conducted along the entire Pacific coast of Mexico to estimate the size of the leatherback nesting population. Surveys had been carried out previously on major leatherback nesting beaches, including annual surveys at Playa Mexiquillo in Michoacan since the 1982-83 season. But these lacked the scope and rigor of the 1995-96 survey. In a detailed account of the 1995-96 survey, Sarti et al. (1996a) reported a total of 4,757 leatherback nests deposited along the Pacific coast of Mexico, implying nesting activity by 898 nesters (assuming an average of 5.3 nests per nester). In a summary report of the survey Sarti et al. (1996b) published a somewhat higher figure of 5,222 nests, equivalent to 985 nesters.

In any event, results of the nesting population survey in 1995-96 are consistent with results of earlier surveys, particularly at Mexiquillo beach, where survey data show roughly a 10-fold decline in the abundance of leatherback nesters during the last decade (Fig. 10; estimated from data given in Sarti et al., 1996b).

The recent survey results are even more alarming when compared with much higher estimates of nesting during the 1980-81 nesting season reported by Pritchard (1982). Pritchard roughly estimated that 30,000 leatherbacks nested during the 1980-81 season. To derive this estimate Pritchard made three key assumptions: (1) each night during the peak of the nesting season about 500 nesters were ashore at Tierra Colorada, Guerrero, one of the principal nesting beaches; (2) nesting at this beach accounted for about one-third of all Pacific leatherback nesting in Mexico; and (3) on an average night during the peak of the nesting season about 5% of the season's total number of nesters would be expected to emerge. Pritchard called his computations "preliminary" and admitted that they could be significantly inaccurate due to compounding of uncertainties. It may be impossible to fully reconcile Pritchard's estimate of 30,000 nesters with current survey results. Obstacles include the lack of comprehensive survey information at other beaches besides Mexiquillo (until the 1995-96 survey); an apparent lack of fidelity to nesting beaches by remigrant nesters; and unknown levels of mortality to leatherbacks from egg and nester harvests, fishing, and other sources.

One approach to estimating the total Mexican Pacific leatherback nesting population for recent years is to scale up the annual results at Mexiquillo beach, assuming the geographical distribution of nesting activity has been constant. In the comprehensive 1995-96 survey about 14.4% of the nesting activity was estimated to have occurred at Playa Mexiquillo (Sarti et al., 1996a). Thus, a raising factor of 6.9 can be applied to the data in Fig. 15 to estimate a time series of the total, coastwide nesting population. A simple backward projection of this time

series then suggests that in 1980-81 the entire Pacific nesting population of Mexican leatherbacks was probably about 6,000-8,000 turtles, far fewer than Pritchard estimated.

Egg and Nester Mortality

The taking of eggs and killing of nesting females by local villagers have been widely acknowledged as factors contributing to the severe decline in the Mexican leatherback nesting population (Eckert, 1993). In general, egg and nester harvests have been poorly documented in Mexico, at least in the English-language literature I have examined. This may be particularly true since such practices were banned in 1990 (Aridjis, 1990). Pritchard (1982) reported that "many hundreds" of nesters were killed each year in Mexico's Pacific leatherback population and that in areas where killing of adults was rare, extensive egg harvesting nevertheless occurred.

Incidental Fishing Mortality

Mexican leatherbacks are caught in fishing gear deployed in Pacific coastal waters of Mexico (generally undocumented), in various fisheries operating off the coasts of other countries and on the high seas. MtDNA analyses of tissue biopsies collected by NMFS observers indicate that at least some of the leatherbacks taken in the Hawaii-based longline fishery may have come from a Mexican rookery (Dutton et al., 1997; see earlier discussion).

Adult female leatherbacks tagged on Mexican nesting beaches have been caught in waters off Chile by purse seines and gill nets (Marquez and Villanueva, 1993). In this region a major Chilean fishery for swordfish operates (Frazier and Brito Montero, 1990). The artisanal gillnet fishery and industrial longline fishery for swordfish expanded greatly during the early 1980s (Yáñez et al., 1997; in press). Although there has been no systematic monitoring of turtle takes in these fisheries, Chilean sources surmise that the total annual take may be quite large, (e.g., "several hundred per year"; Frazier and Brito Montero, 1990). An annual leatherback kill of this magnitude may well have occurred during the height of the swordfish fishery 5 years ago. There is considerable uncertainty here. Data on total driftnet and longline effort are fairly complete, but information on take per unit of effort is meager and largely anecdotal. Much better data are required to compute a reliable estimate of total kill.

Sustainable Levels of Egg and Nester Mortality

The impacts of egg and nester harvests on the Mexican leatherback population cannot be fully assessed until accurate harvest data are assembled and made available. This may never be possible. Meanwhile, it is instructive to see what the impacts might be hypothetically given assumptions about life history. Based on our best, current understanding of Mexican leatherback growth, natural mortality, maturation and reproduction I used TURTSIM to

explore the joint effects of egg harvest and nester mortality on the equilibrium nester abundance.

The simulations assumed that there was no incidental mortality in the pelagic stages due to fishing or other causes and that with only natural mortality operating the equilibrium annual nesting population would be 10,000 females. Under these conditions, annual survival during the P1 pelagic stage was estimated at 51.2% during TURTSIM initialization. The analysis showed that the leatherback population would collapse if either the egg harvest rate exceeded about 36% per year or the nester harvest rate exceeded about 9% per year on a perpetual basis (Figs. 7 and 9). Clearly, an annual nester kill of "many hundreds" (Pritchard's estimate of 1980 conditions) along with extensive egg harvesting would appear to be unsustainable. The equilibrium relationships can be used to identify ranges of sustainable rates of egg and nester mortality under various conservative recovery objectives, e.g., rebuilding the nesting population to 60% or 75% of the maximum level (Figs. 9 and 11).

Given high rates of egg harvest and nester mortality (e.g., those likely to have prevailed before the Presidential ban in 1990) the simulated nesting population is predicted to decline rapidly, in a manner consistent with the steep decline observed over the past decade at Playa Mexiquillo. In particular, seemingly minor rates of nester harvest cause a sharp decline in the expected equilibrium level of nesters (Fig. 7). The TURTSIM analysis shows that without nester kills, low levels of egg take may be compensated to some extent by density-dependent responses of the population, but egg harvest rates only slightly higher may eventually lead to a collapse of the population. These are the expected results even if all incidental fishing mortality to pelagic stage turtles is eliminated.

The shape of the equilibrium curves is altered if it is assumed that density dependence in growth and remigration rates is linear (i.e., $\beta = 1$) (Fig. 7). Further, under the linearity assumption the sustainable rates of egg and nester mortality are reduced (Fig. 8).

Sustainable Levels of Fishing Mortality

The TURTSIM leatherback model indicates that a fishing mortality coefficient up to about 0.019 per year may be compensated by density-dependent changes in the population if there is no harvest of eggs or nesters (Fig. 11). This would allow a maximum sustainable incidental mortality of about 1,400 turtles per year (P2 juveniles and adults). If fishing mortality is increased further, however, equilibrium levels of nester abundance and sustainable take fall off sharply under the conditions assumed. Fishing mortality coefficients above about 0.024 per year are not sustainable. A relative nester abundance of 75% maximum is associated with an incidental fishing mortality of 0.016.

Even modest levels of egg and nester harvest may greatly reduce the maximum sustainable level of incidental fishing mortality, however. For example, in the TURTSIM model simulations a 5% egg take and 2% nester kill each year reduced the upper limit of sustainable fishing mortality coefficients by 40% and the maximum sustainable fishing mortality to no more than about 800 turtles per year (Fig. 11). Because of the assumed density-dependent compensation the simulated population was able to accommodate low levels of incidental fishing mortality. But once maximum growth and remigration probabilities were reached equilibrium levels of sustainable mortality and nester abundance dropped off steeply with increases in the fishing mortality coefficient.

Projections of Hawaii Longline Fishery Impacts

TURTSIM was used to assess impacts of the Hawaii longline fishery under six scenarios. In each scenario, it was assumed that all leatherbacks killed in the Hawaii longline fishery were from rookeries in Mexico, and the instantaneous mortality rate applied by the Hawaii fishery was equivalent to the upper bound of the 95% confidence range for mortality, as estimated by Skillman and Kleiber (in prep.). In Scenarios 1-3, it was assumed that the only human-induced mortality after 1997 was caused by the Hawaii longline fishery. Other conditions assumed were:

Scenario 1 - All incidental mortality from Hawaii longline fishing was eliminated after 1997, so that only natural mortality remained.

Scenario 2 - The Hawaii longline fishery continued to operate at current (1997) level of intensity, exerting the same coefficient of incidental fishing mortality on Mexican leatherbacks.

Scenario 3 - The coefficient of incidental fishing mortality on Mexican leatherbacks generated by the Hawaii longline fishery expanded 5-fold over the current level (due to sudden expansion of fishing effort, increased vulnerability of leatherbacks, or other factors).

In **Scenarios 4-6** the same conditions were assumed with respect to the Hawaii longline fishery as in Scenarios 1-3, but other human-induced mortality was added. Specifically, it was assumed that other pelagic fishing mortality occurred each year at a level 10 times the current (1997) intensity of the Hawaii longline fishery; 5% of the eggs were harvested annually, and 2% of the nesters killed.

To set up the projections, the current population abundance and stage structure were estimated by supposing an equilibrium abundance of 10,000 nesters in 1950, then running TURTSIM forward in time while applying an assumed history of human-induced mortality. The additional sources of mortality included egg and nester harvest, high-seas driftnet fishing (1980-92), Hawaii longline fishing, and other (unspecified) mortality to the P2 and adult stages. Levels of Hawaii longline mortality and high-seas

driftnet mortality were set at values estimated from observer data (see Table 5). Levels of mortality for other sources, however, were unknown and were adjusted arbitrarily to force TURTSIM to mimic the steep decline apparent in the Mexiquillo nesting data. Under these constraints, very high levels of egg harvest (up to 90%) and nester mortality (up to 20%), and other pelagic fishing mortality (over 1,000 P2 juveniles and adults per year) were required to cause such a precipitous decline in the nesting population.

Under Scenarios 1-3, the simulated nesting population was predicted to recover slowly. In projections over the next half century or so there was no discernable difference in the expected nester abundance under the assumed current level of fishing intensity and the hypothetical case with zero mortality in Hawaii longline gear (Fig. 13). Under Scenarios 4-6 the nesting population was projected to remain at about its current level over the next several decades (Fig. 13).

ESTIMATING UPPER LIMITS FOR ALLOWABLE INCIDENTAL FISHING MORTALITY

Under current practice in the U.S., allowable take and mortality levels are upper thresholds for monitoring fishery interactions. When either threshold is exceeded an internal NMFS consultation is triggered under Section 7 of the Endangered Species Act. Presumably, the allowable take and mortality levels must not exceed maximum levels required to achieve recovery goals for the population. As discussed later, setting upper mortality limits should take into consideration turtle population recovery objectives, assessments of the population's size and dynamics, and the effects of other human activity (including other fishing mortality). Allowance should be made for uncertainty in the population assessment and other factors. Computer simulation studies, using TURTSIM or other programs, can provide guidance on setting such upper limits. Presented below are results from TURTSIM simulations leading to estimates of maximum allowable incidental mortality for Japanese loggerheads and Pacific leatherbacks. Two methods were applied. The first method is the PBR ("Potential Biological Removal") formula now widely used in marine mammal conservation. PBR is a robust method particularly useful as a "rule-of-thumb" (including default values of key parameters) when data to support complex models are unavailable. Wade (in press) provides a definitive account of PBR development. The second method explored here is closely akin to PBR and is dubbed MAIM ("Maximum Allowable Incidental Mortality").

Computing Potential Biological Removal (PBR)

The potential utility of the PBR method for estimating maximum allowable incidental turtle mortality levels was first pointed out by Gerrodette (1996). NMFS has subsequently funded research to

develop a PBR-like method applicable to sea turtles, tailored to the severe data limitations typically faced in sea turtle recovery plans (Selina Heppell, pers. commun.).

The PBR is an estimate of the maximum incidental take that could be allowed in a defined time interval (per year, in our case) while enabling a depleted population to recover to a higher, target level of abundance. The Marine Mammal Protection Act (as amended in 1994) stipulates that PBR be applied to assure recovery of a population to its "optimum sustainable population" size. According to Wade (in press), this is typically about 50-70% of the carrying capacity, the maximum equilibrium population size in the absence of human impacts. The PBR is computed as (Gerrodette, 1996; with modified notation):

$$\text{PBR} = N \times (F_{\max}/2) \times R$$

where N is a conservative estimate of the current abundance (e.g., the 20th percentile of a log-normally distributed population estimate), F_{\max} is the maximum net per capita population growth rate (e.g., the net growth rate at very low population levels), and the scalar R is the so-called "recovery factor" set at 0.50 for populations classified as "threatened" and 0.10 for populations considered "endangered." The recovery factor provides a degree of hedging against uncertainty about the population's size and productivity. Accordingly, the PBR specifies a conservative level of allowable incidental mortality given current conditions in the population. As these conditions change or estimates of them change over time, PBR is adjusted appropriately. The PBR approach is attractive because it (1) provides a safe path to population recovery; (2) involves just a few, understandable parameters; and (3) is adaptive with respect to changes in both population status and information quality.

The TURTSIM simulations provide necessary input parameters for PBR estimates. Using loggerheads as an example, Fig. 4 shows the relationship between the coefficient of fishing mortality and equilibrium levels of relative nester abundance. It also shows the number of P2 stage loggerheads that would be killed, under equilibrium conditions, at each coefficient of incidental fishing mortality, assuming a maximum nesting population of 10,000. The simulations leading to the graph assumed that there was no other human-induced mortality in effect, e.g., no egg harvest or nester harvest. The equilibrium conditions imply that the incidental P2 mortality just balances the net increase in P2 juveniles which would otherwise occur under these conditions. Thus, under equilibrium conditions the coefficient of P2 fishing mortality is equivalent to the net per capita growth rate in P2 juvenile loggerheads. From Fig. 4, an estimate of the PBR parameter F_{\max} is, therefore, about 0.046 per year. From the same set of simulations, an estimate of the equilibrium abundance of P2 juvenile turtles corresponding to the current equilibrium level of 1,000 nesters is about 28,000 turtles. Thus, for Japanese loggerheads we have the estimate (Table 6): $\text{PBR} = 28,000 \times (0.046/2) \times 0.5 = 318 \text{ P2}$

juveniles. Here, as in the Workshop report, we have assumed that the actual loggerhead nester abundance is roughly in equilibrium.

If the form and magnitude of density dependence are as assumed in the TURTSIM simulations, the PBR formula appears to give a more conservative guideline than intended, i.e., the nesting population would be returned to about 90% of carrying capacity rather than about 60%. Moreover, the 60% target is beyond the region of maximum sustainable incidental mortality (surplus production), suggesting that a higher target, perhaps 75% or so, would be preferable. This consideration leads us to the MAIM procedure, to be described below.

Similar steps lead to PBR estimates for Pacific leatherbacks. In dealing with leatherbacks, I computed a combined estimate for all populations based on the estimated aggregate nester abundance. Best current estimates indicate an aggregate nester abundance of about 5,000 leatherbacks, as follows:

<u>Population</u>	<u>Nesters</u>	<u>Source</u>
Mexico	985	Sarti et al. (1996b)
Costa Rica	1,240	Spotilla (1996)
Panama	20	Spotilla (1996)
Malaysia	25	Chan and Liew (1996)
Irian Jaya	2,600	Suwelo et al. (1994)
TOTAL	4,870	

This total covers the principal nesting populations, but omits several populations with minor or unknown nesting (e.g., China, Solomon Is., Papua New Guinea). Based on the Mexican leatherback simulations, the total nesting figure may be multiplied by a factor of 16-18 to estimate a current abundance of P2 and adult turtles of about 85,000. I assumed that the population dynamics of Mexican leatherbacks were representative of other Pacific populations. Based on an estimated maximum net per capita growth rate of P2 and adult leatherbacks in the Mexican population, the aggregate estimate of Pacific leatherback PBR (Table 6) is $PBR = 85,000 \times (0.024/2) \times 0.1 = 102$ P2 juveniles and adults.

Although green turtles appear to be little affected by the Hawaii longline fishery, an estimate of PBR was also computed for Hawaiian green turtles (Table 6) assuming a current statewide population of about 700 nesters and a maximum historic nester abundance of 5,000. TURTSIM simulations suggest that the corresponding abundance of vulnerable pelagic stage greens is 54,000 turtles. The estimated maximum sustainable levels of fishing mortality for Hawaii green turtles is considerably higher than for the other populations considered above (Fig. 14). This is due to several factors, including the small size of green turtles taken in the fishery relative to size at maturity, and a relatively high reproductive success of the green turtle population. In regard to reproductive success, TURTSIM estimates the equilibrium number of post-hatchlings produced for each female reaching

maturity (a post-hatchling here defined as a turtle surviving one week after emergence):

<u>Population</u>	<u>Post-Hatchlings Per Recruit</u>
Hawaiian green turtle	708
Japanese loggerhead	298
Malaysian leatherback	733
Mexican leatherback	175

No assessments have been made of other green turtle populations potentially affected by the Hawaii longline fishery, e.g., those in Mexico or Japan.

Computing Maximum Allowable Incidental Mortality (MAIM)

Computations for MAIM estimates are similar to those for PBR. As in the PBR approach, it is assumed the population would increase over time to a maximum average level of abundance if all human impacts were eliminated and that the recovery objective is to rebuild the population to an acceptable fraction of the maximum population level. The difference is that in computing MAIM we multiply the current population size by the coefficient of incidental fishing mortality expected to produce the desired degree of recovery, whereas PBR always (and conservatively) assumes the "optimal" incidental mortality rate and is equal to half the maximum net per capita growth rate. If density dependence is linear (i.e., has the form based on $\beta = 1$ in Fig. 1) PBR and MAIM should be approximately the same. The MAIM approach involves several steps:

1. Derive the equilibrium relationship between the relative abundance of nesters (as a proportion of maximum nesters) and the coefficient of incidental fishing mortality, assuming the same size selectivity as in the Hawaii longline fishery.
2. Choose the recovery objective with respect to the equilibrium relative nester abundance, then find the associated optimum incidental fishing mortality coefficient, denoted by F_{opt} .
3. Estimate the current abundance of pelagic turtles vulnerable to incidental capture in the Hawaii longline fishery (the vulnerable size group), N .
4. Set the value of the "recovery factor", R . Provisionally, use the same default values as in the PBR method.
5. Compute the maximum allowable incidental mortality, MAIM, as the product of these elements:

$$MAIM = N \times F_{opt} \times R$$

The MAIM procedure was applied to Japanese loggerheads, Pacific leatherbacks, and Hawaiian green turtles (Table 6). A recovery goal of 75% maximum nester abundance was arbitrarily chosen; thus $F_{opt} = F_{0.75}$. In each case, MAIM exceeds PBR, due to assumptions about density dependence of growth and remigration rates. That is, with $\beta = 10$, $F_{0.75} > F_{max}/2$.

Pacific leatherback results show clearly the effect of egg and nester harvests on the maximum allowable incidental fishing mortality (Table 6); both MAIM and PBR estimates are greatly reduced when other human impacts are assumed to occur. Clearly, the proper setting of incidental mortality levels will require a consideration of, and allowance for, all human impacts. Estimates of allowable mortality in Table 6 are strictly conditional. In order to fully account for all sources of human-caused mortality, establish allowable total mortality levels consistent with agreed recovery goals, and determine appropriate and fair allocations of incidental mortality among sources; international cooperation will have to be significantly improved.

SUMMARY OF IMPACTS

Japanese Loggerheads

The revised TURTSIM analysis of Japanese loggerheads, like the Workshop assessment of this population, indicates that the current impact of the Hawaii longline fishery is relatively minor. Based on a computed current abundance of 28,000 stage P2 (late pelagic) turtles, the estimated annual incidental mortality of 50-60 loggerheads caused by the fishery translates to a mortality coefficient of about $2.0 \times 10^{-3} \text{ yr}^{-1}$. This is only about 5% of the total incidental mortality coefficient of stage P2 turtles consistent with recovery of the nesting population to 75% of its pristine abundance, assuming there is no human-induced mortality to other size classes. The full impact of the Hawaii fishery cannot be determined until a complete assessment is made of other human-induced mortality, including mortality in other fisheries.

As noted in the Workshop report, the status of loggerhead nesting in Japan is not clear. Time series of recorded nesting levels are available for principal nesting beaches, but some series are quite short. Survey methods and estimation procedures have not been described, so it is not possible to evaluate survey results and potential biases. Except for some understanding of nesting biology and juvenile growth rates, little is known about life history and population parameters of Japanese loggerheads. Of particular concern is the lack of comprehensive data on other incidental mortality sources. More international cooperation is required to increase reliability of the population assessment, document mortality from all sources, and establish comprehensive guidelines for allowable incidental mortality.

Pacific Leatherbacks

Similarly, the TURTSIM analysis of Pacific leatherbacks indicates that the current impact of the Hawaii longline fishery is relatively minor. Under current conditions, the estimated annual incidental mortality of about 10 leatherbacks caused by the fishery translates to a mortality coefficient of about $1.0 \times 10^{-4} \text{ yr}^{-1}$ with respect to the aggregate population of P2 and adult leatherbacks, assuming a current abundance of 85,000 P2 and adult turtles. This is less than 1% of the total incidental mortality coefficient in these size classes consistent with recovery of the aggregate nesting population to 75% of its pristine abundance, provided humans do not cause mortality in other developmental stages (e.g., through egg harvests, nester kills, or incidental fishing mortality of the P1 stage juveniles). The full impact of the Hawaii longline fishery on leatherbacks cannot be assessed until the magnitude of other fishing mortality, egg takes, and nester kills are determined and population parameters are better known. Despite recent measures to curb leatherback mortality on nesting beaches in Mexico and other countries, the cumulative mortalities from human activities, including coastal and high-seas fishing, may still be excessive.

Additional uncertainty about the Hawaii longline impacts stems from lack of reliable data on stock structure and natal origins of leatherbacks caught in the fishery. While preliminary mtDNA work suggests a mixture of stocks are involved, a much larger sample will be required for a definitive assessment. Once better data on stock structure are available, the assessment of leatherback population status and fishery impacts should be done on a population-specific basis.

SETTING GUIDELINES FOR ALLOWABLE INCIDENTAL MORTALITY

The Ideal Situation: Multinational Recovery Plans

In many fisheries, the setting of incidental mortality limits for turtles has lacked a rigorous, quantitative rationale. Ideally, such a rationale would be provided by a comprehensive recovery plan developed cooperatively by concerned governments. While such a plan could take various forms, we might envision it to include several key elements:

1. Definition of recovery objectives.
2. Identification of factors detrimental to population recovery and steps to mitigate them.
3. Comprehensive strategy for monitoring population abundance and mortality (e.g., nesting beach surveys and observer programs to monitor incidental fishing mortality).

4. Recovery management decision procedures, e.g., procedures for setting, allocating and adjusting incidental mortality levels.
5. Criteria to judge whether management actions are having the desired effects on population recovery.

Thus far no multinational recovery plans exist. In lieu of them, however, national or regional recovery plans have been developed to identify recovery actions that can be taken locally, pending development of full-scale, multinational plans. In the case of turtles affected by the Hawaii longline fishery, recovery plans for green turtles, hawksbills, olive ridleys, loggerheads and leatherbacks have been drafted by NMFS and USFWS. These are U.S. plans only; they implement no multinational commitments or agreements. Further, a draft of U.S. plans for the Pacific do not yet include many of the five elements above.

In the U.S., recovery plans themselves are part of a recovery process that starts with listing of the species or population and ends when delisting is accomplished. The process implies some criteria for deciding when to alter the listed status of a population, e.g., a set of threshold levels of abundance or other reference points. A population depleted below a lower critical level may be initially listed as "endangered," for example, then uplisted to "threatened" once it rebuilds to higher levels, and finally delisted when an upper target threshold is reached (Fig. 15; in this example recovery is judged by status of the nesting population). Recovery management plans ought to include systematic monitoring of progress and decision mechanisms (analogous to Section 7 consultations) for assessing new information and, if appropriate, adjusting allowable incidental mortality levels.

Interim Incidental Mortality Guidelines

NMFS should develop a comprehensive quantitative framework for turtle population recovery management decisions. This should incorporate models of turtle population dynamics, recovery goals, monitoring schemes and decision criteria. The research funded by F/PR to develop a PBR-like procedure for turtles should result in a broadly applicable quantitative approach to setting total allowable mortality limits. Once developed, such a framework could be adopted unilaterally and applied within the context of U.S. recovery plans while efforts are accelerated to gain the support and data required for multilateral plans.

Because a quantitative framework has not been available, NMFS has, heretofore, adopted interim guidelines for the Hawaii longline fishery. The current mortality limits were set provisionally at the average or expected level of mortality given prevailing or projected levels of fishing activity, under the presumption that such mortality, taken alone, was not placing the populations in jeopardy. It appears that similar approaches have been used provisionally in other fisheries (e.g., Atlantic and Gulf of Mexico

pelagic longline and driftnet fisheries; Gulf of Mexico shrimp trawl fishery). In these status quo approaches, there is no explicit linkage between the mortality limits and a specified quantitative recovery objective (e.g., a higher populations size). Further, the joint and cumulative impacts of human activities may not be fully considered.

Here, two ways to set guidelines for incidental turtle mortality in the Hawaii longline fishery are described. The purpose of the guidelines is to define a mechanism for triggering a Section 7 Consultation when incidental mortality is excessive. The methods assume that turtle interactions are monitored annually through a sample survey of fishing operations (observer program) leading to a variety of statistics on the mortality level, including Bayesian posterior probability distributions of the total mortality level and related estimates of quantiles and confidence ranges.

Alternative A: Assess risk of exceeding allowable mortality.

This (preferred) approach explicitly considers the total allowable incidental mortality, an allocation of mortality to the Hawaii fishery, and monitoring of the actual mortality within a specified margin of monitoring risk. Four steps are involved:

1. Estimate an overall allowable mortality (e.g., PBR), then allocate a reasonable share of this to the Hawaii longline fishery; call this allowable share of the mortality H .
2. Specify an acceptable monitoring risk, the probability that the incidental fishery mortality will exceed H in a given year; call this $R(H)$.
3. Each year, estimate the posterior distribution of Hawaii longline incidental mortality, based on prior information and newly acquired observer data and logbook statistics, and from it compute the posterior probability that the mortality would have exceeded H ; call this $P(H)$, the monitoring risk.
4. If $P(H) > R(H)$ open a Section 7 consultation.

Alternative B: Assess odds of departing from current norm.

The second alternative is a status quo approach in that it allows the fishery to continue operations within its normal scope of variability, with the proviso that the condition of the turtle population is explicitly considered; i.e., there are assurances that the allowed incidental mortality will not jeopardize the turtle population. Five steps are involved:

1. Given accumulated data on levels of fishery interactions and mortality in the Hawaii longline fishery, derive the current "norm" probability distribution of annual

incidental mortality. Compute associated quantiles, including the current median mortality.

2. Specify a threshold level of median mortality, H , higher than the current median mortality, above which a significant departure from the norm will be indicated.
3. Provide evidence that H is safe (e.g., well below estimates of PBR).
4. Incorporating the current year's data, estimate an updated (posterior) mortality distribution, and use this distribution to estimate $P(H)$, the probability that the mortality caused by the fishery would have exceeded H .
5. If $P(H) > 0.5$ open a Section 7 consultation.

These procedures differ from the one currently used by recognizing the variability inherent in processes that determine the actual level of incidental mortality and mortality estimates. Managers may be able to control the level and configuration of nominal effort (e.g., fleet size and composition), but the mortality generated by the fishery will be influenced by random variation in many factors. Prominent among these would be the distribution and local abundance of turtles, spatio-temporal patterns of longline effort, vulnerability of turtles to fishing gear, frontal structure and other oceanographic conditions. Estimates of incidental mortality distributions and important derivative statistics, e.g., quantiles, will also be affected by sampling variation in observer data. Monitoring decisions must take into consideration both kinds of uncertainty, inherent natural process variation and sampling variation. The alternative procedures also involve a broader scope of inference than the current approach, focusing attention primarily on the statistical distribution of annual mortality caused by the fishery as it is presently constituted, not just the mortality level realized in the current year.

In Alternative A, the monitoring risk clearly affects the biological risk to the population. Ideally, the biological risk will have been incorporated into estimation of the PBR or MAIM. Biological risk can, of course, be reduced by implementing mortality mitigation measures (e.g., cutting back fishing effort, altering fishing gear or methods, lowering mortality from other sources). For a fixed value of H , the monitoring risk can be reduced by decreasing uncertainty in the annual take assessment, i.e., by increasing observer program coverage.

Alternative A cannot be effectively implemented until procedures for setting mortality guidelines (PBR, MAIM, or other methods) and allocating the allowable incidental mortality among sources are developed and agreed upon. The allocation step is especially problematic because mortalities of unknown magnitude are occurring through numerous human activities, including other fisheries (besides the Hawaii longline fishery). Alternative B, or

similar methods, may be used on an interim basis until more sophisticated approaches are established. The threshold level of median mortality in Alternative B is, of course, subjective. It would be prudent to set H lower for severely reduced populations and for a source of mortality that is thought to account for a large fraction of the total incidental mortality; a higher level of H would be more tolerable for healthier populations and a less onerous source of mortality.

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Table 1. Assumed life history characteristics and parameters for Japanese loggerheads.

Stage Structure:	
<u>Developmental Stage</u>	<u>Length (SCL, cm)</u>
Early pelagic (P1)	5 - 45
Late pelagic (P2)	46 - 75
Benthic immature (B)	76 - 84
Adult (A)	>= 85
Stage-specific Base Annual Survival Probability:	
<u>Stage</u>	<u>Survival</u>
P1	estimated by TURTSIM at 0.79
P2	0.85
B	0.88
A	0.95
Reproduction:	
Clutches/nester	3.4
Eggs/clutch	112
Probability survival from egg to P1 stage	0.138
Hatchling sex ratio (females:males)	1:1
Maturation Ogive:	
<u>Probability Turtle Matures in Current Year</u>	<u>Corresponding Length (SCL, cm)</u>
0	79
25	82
50	84
75	86
100	89
Growth:	
Assumed Mean Age at Maturation (yrs)	25
von Bertalanffy L_{inf} (SCL, cm)	105
von Bertalanffy K (per year)	0.061 at current population density 0.055 at maximum population density
Percent Adult Females Nesting:	20 at current population density

Table 2. Assumed life history characteristics and parameters for Malaysian leatherback population.

Stage Structure:	
<u>Developmental Stage</u>	<u>Length (SCL, cm)</u>
Early pelagic (P1)	6 - 74
Late pelagic (P2)	75 - 144
Adult (A)	>= 145

Stage-specific Base Annual Survival Probability:	
<u>Stage</u>	<u>Survival</u>
P1	estimated by TURTSIM at 0.39
P2	0.85
A	0.95

Reproduction:	
Clutches/nester	5.7
Eggs/clutch	85
Probability survival from egg to P1 stage	0.267
Hatchling sex ratio (females:males)	1:1 (natural)

Maturation Ogive:	
<u>Probability Turtle Matures in Current Year</u>	<u>Corresponding Length (SCL,cm)</u>
0	130
25	140
50	145
75	150
100	160

Growth:	
Assumed Mean Age at Maturation (yrs)	14
von Bertalanffy L_{inf} (SCL, cm)	170
von Bertalanffy K (per year)	0.128 at current population density 0.116 at maximum population density

Percent Adult Females Nesting:	44 at current population density
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Table 3. Assumed life history characteristics and parameters for Mexican leatherback population.

Stage Structure:	
<u>Developmental Stage</u>	<u>Length (SCL, cm)</u>
Early pelagic (P1)	6 - 74
Late pelagic (P2)	75 - 144
Adult (A)	>= 145
 Stage-specific Base Annual Survival Probability:	
<u>Stage</u>	<u>Survival</u>
P1	estimated by TURTSIM at 0.51
P2	0.85
A	0.95
 Reproduction:	
Clutches/nester	5.3
Eggs/clutch	62.7
Probability survival from egg to P1 stage	0.093
Hatchling sex ratio (females:males)	1.7:1
 Maturation Ogive:	
<u>Probability Turtle Matures in Current Year</u>	<u>Corresponding Length (SCL,cm)</u>
0	130
25	140
50	145
75	150
100	160
 Growth:	
Assumed Mean Age at Maturation (yrs)	14
von Bertalanffy L_{inf} (SCL, cm)	170
von Bertalanffy K (per year)	0.128 at current population density 0.116 at maximum population density
Percent Adult Females Nesting:	44 at current population density

Table 4. Assumed life history characteristics and parameters for the Hawaiian green turtle population.

Stage Structure:	
<u>Developmental Stage</u>	<u>Length (SCL, cm)</u>
Early pelagic (P1)	5 - 19
Late pelagic (P2)	20 - 34
Early juvenile (J1)	35 - 50
Late juvenile (J2)	51 - 69
Subadult (B)	70 - 84
Adult (A)	>= 85
Stage-specific Base Annual Survival Probability:	
<u>Stage</u>	<u>Survival</u>
P1 & P2	estimated by TURTSIM at 0.66
J1	0.75
J2	0.85
B	0.90
A	0.95
Reproduction:	
Clutches/nester	3.7
Eggs/clutch	104
Probability survival from egg to P1 stage	0.33
Hatchling sex ratio (females:males)	1:1
Maturation Ogive:	
<u>Probability Turtle Matures in Current Year</u>	<u>Corresponding Length (SCL, cm)</u>
0	81
25	84
50	86
75	88
100	91
Growth:	
Assumed Mean Age at Maturation (yrs)	25
von Bertalanffy L_{inf} (SCL, cm)	105
von Bertalanffy K (per year)	0.065 at current population density 0.059 at maximum population density
Percent Adult Females Nesting:	33 at current population density

Table 5. Estimated takes and mortalities of sea turtles incidental to fishing in the North Pacific Ocean.

Fishery	Year	Species	Estimated Annual Take		Estimated Mortality		Sources
North Pacific high-seas driftnet: squid, tuna & billfish	June 1990-May 1991	Loggerhead	2,986		805		International observer program data, fleet effort statistics; Wetherall et al. (unpub. ms.)
		Leatherback	1,002		111		
		Green	378		93		
		Olive ridley	0		0		
		Hawksbill	7		2		
		TOTAL	4,373		1,011		
Hawaii longline: swordfish & tuna	1994	Loggerhead	(A) 207	(B) 301	(A) 31	(B) 51	NMFS observer program data, fleet effort statistics (logbooks); (A)Diaz-Soltero (1996); (B)Skillman and Kleiber (in prep.) ★
		Leatherback	122	132	18	9	
		Green	34	15	5	1	
		Olive ridley	78	120	12	32	
		Hawksbill	0	0	0	0	
		TOTAL	441	568	67	93	
		1995	Loggerhead	413	339	62	
	Leatherback		81	156	12	11	
	Green		0	41	0	1	
	Olive ridley		81	124	12	33	
	Hawksbill		0	0	0	0	
	TOTAL		575	660	87	102	
	1996	Loggerhead		358		60	
		Leatherback		159		11	
		Green		42		1	
		Olive ridley		129		34	
		Hawksbill		0		0	
		TOTAL		688		106	

*

(A) and (B) estimates were computed using different methods and assumptions. The (A) take estimates were derived separately for each year in accordance with the pilot stratified survey design developed by DiNardo (1993). The (B) take estimates, using pooled observer data for 1994-1996, were based either on a regression-tree model that predicts average take rate as a function of latitude of fishing (loggerheads) or a simple expansion of overall mean take rate (other species). Methods of estimating mortality rates also differed between (A) and (B).

Table 5. (Continued) Estimated takes and mortalities of sea turtles incidental to fishing in the North Pacific Ocean.

Fishery	Year	Species	Estimated Annual Take	Estimated Mortality	Sources
Eastern tropical Pacific purse seine: yellowfin tuna	1993	TOTAL (mostly olive ridley)	295	N/A	IATTC observer program data, fleet catch statistics (logbooks); IATTC 1997.
	1994	TOTAL (mostly olive ridley)	167	N/A	
California drift gillnet: swordfish, thresher shark & mako shark	1990 (last half only)	Leatherback	23	23	NMFS observer program data, fleet effort statistics; Julian and Beeson (in press).
		Loggerhead	0	0	
		TOTAL	23	23	
	1991	Leatherback	10	0	
		Loggerhead	0	0	
		TOTAL	10	0	
	1992	Leatherback	29	15	
		Loggerhead	15	7	
		TOTAL	44	22	
	1993	Leatherback	22	15	
		Loggerhead	37	0	
		Unidentified	22	7	
		TOTAL	81	22	
	1994	Leatherback	6	0	
		Loggerhead	0	0	
		TOTAL	6	0	
	1995	Leatherback	32	26	
		Loggerhead	0	0	
		TOTAL	32	26	

Table 5. (Continued) Estimated takes and mortalities of sea turtles incidental to fishing in the North Pacific Ocean.

Fishery	Year	Species	Estimated Annual Take	Estimated Mortality	Sources
California set gillnet: angel shark & California halibut	1990 (last half only)	Leatherback	0	0	NMFS observer program data, fleet effort statistics; Julian and Beeson (in press).
		Loggerhead	0	0	
		Green/Black	0	0	
		Unidentified	0	0	
		TOTAL	0	0	
	1991	Leatherback	0	0	
		Loggerhead	0	0	
		Green/Black	0	0	
		Unidentified	0	0	
		TOTAL	0	0	
	1992	Leatherback	0	0	
		Loggerhead	8	8	
		Green/Black	8	8	
		Unidentified	0	0	
		TOTAL	16	16	
	1993	Leatherback	0	0	
		Loggerhead	0	0	
		Green/Black	6	6	
		Unidentified	13	6	
		TOTAL	19	12	
	1994	Leatherback	8	8	
		Loggerhead	0	0	
		Green/Black	0	0	
		Unidentified	0	0	
		TOTAL	8	8	

Table 6. Conservative estimates of allowable total pelagic fishing mortality levels for some sea turtle populations involved in the Hawaii longline fishery. Based on TURTSIM simulations. Two methods were used, PBR and MAIM (see text).

Population	Assumed other impacts	Vulnerable length range (CL, cm)	Current abundance (N)	Nominal Incidental Mortality Coefficients		Recovery factor (R)	Maximum Allowable Incidental Mortality	
				$F_{0.75}$	$F_{\max}/2$		MAIM	PBR
Japanese Loggerhead	None	46-75	28,000	0.035	0.023	0.5	504	318
Pacific Leatherback (aggregate)	None	75-170	85,000	0.016	0.012	0.1	136	102
	5% egg take 2% nester kill			0.009	0.008		76	68
Hawaiian Green Turtle	None	20-50	54,000	0.072	0.043	0.5	1,944	1,161

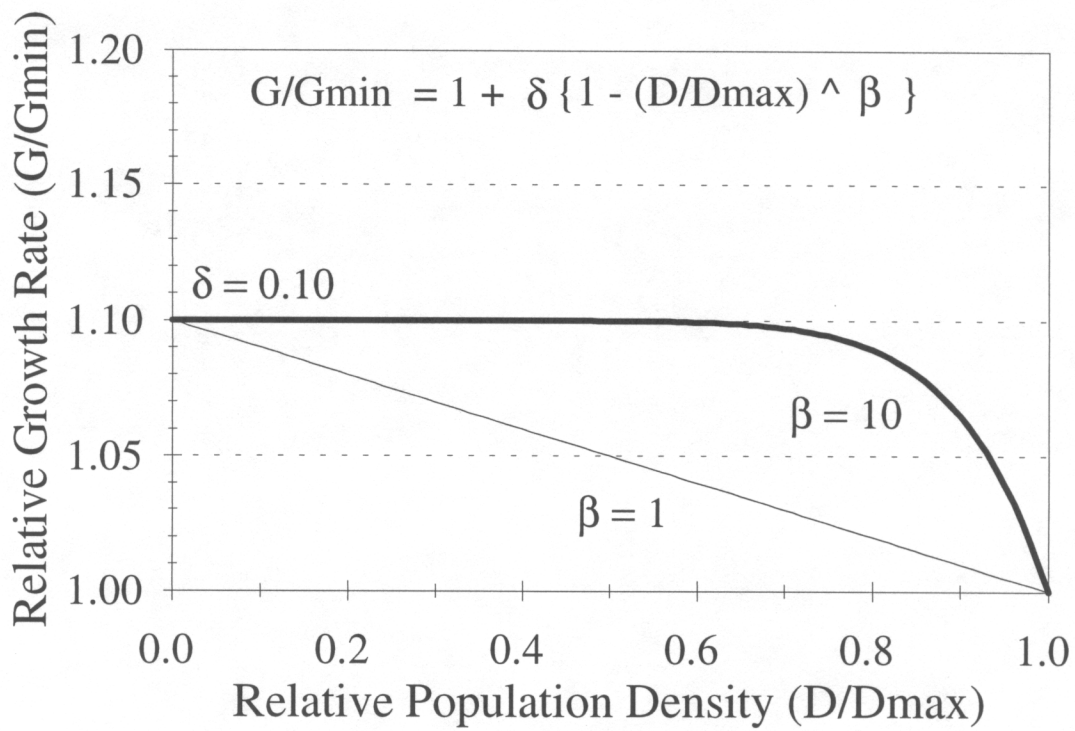


Figure 1. Growth rates observed under current reduced population densities are assumed to be greater than those in effect when turtles were more abundant.

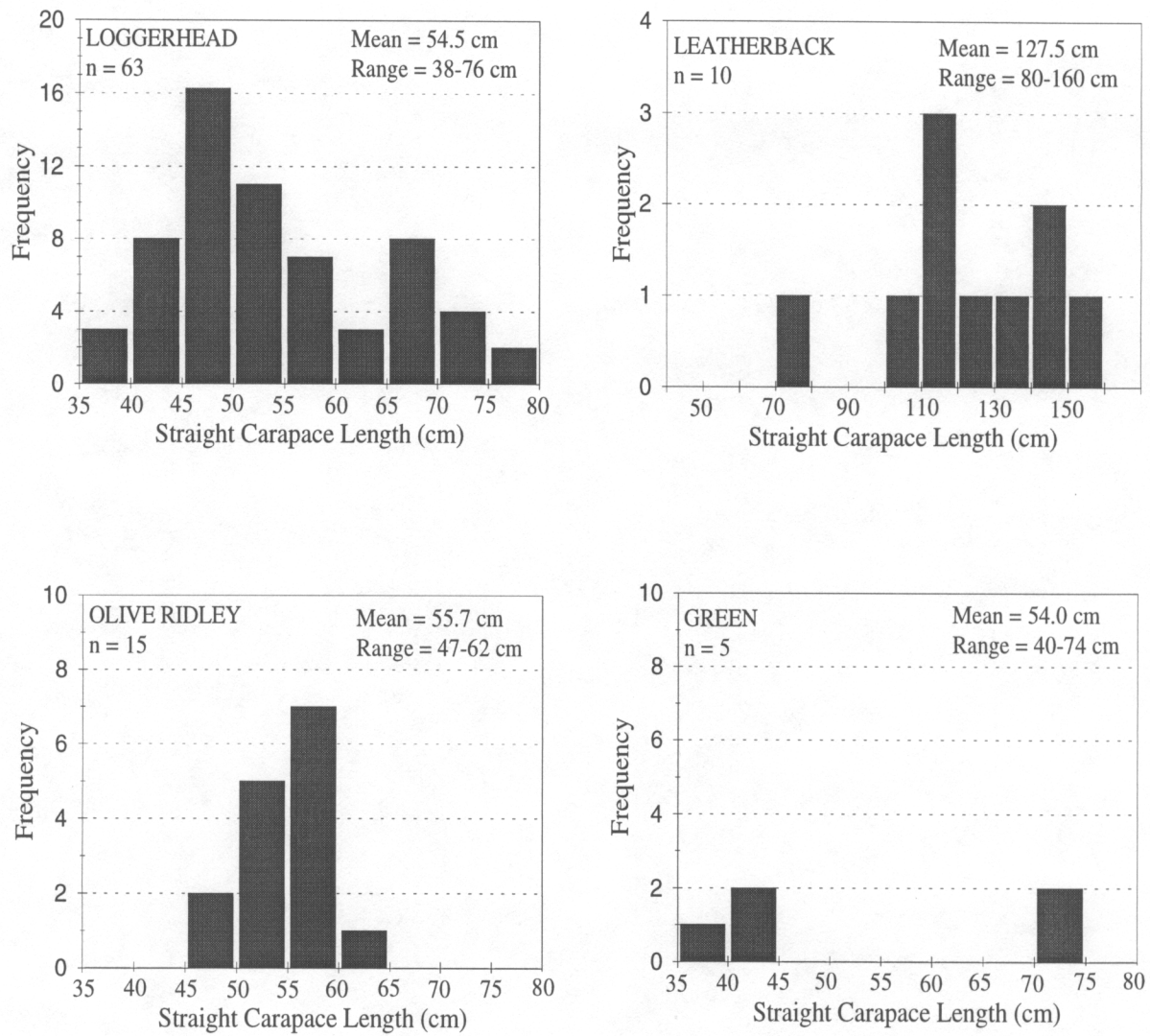


Figure 2. Size frequency distributions of sea turtles taken in the Hawaii-based pelagic longline fishery (1994-present); from NMFS observer data.

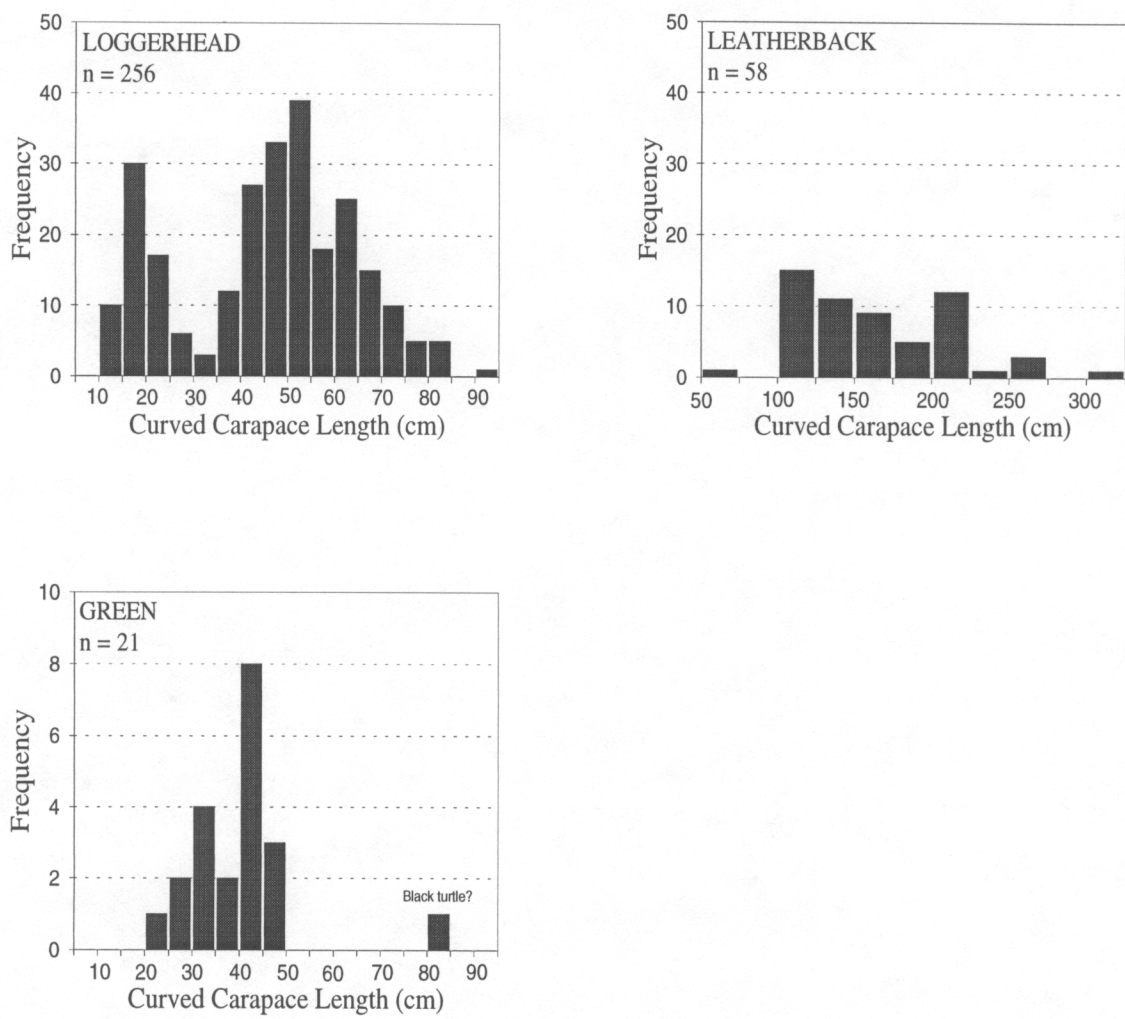


Figure 3. Size frequency distributions of sea turtles taken in the North Pacific high-seas driftnet fisheries during 1990-1991; from observer data.

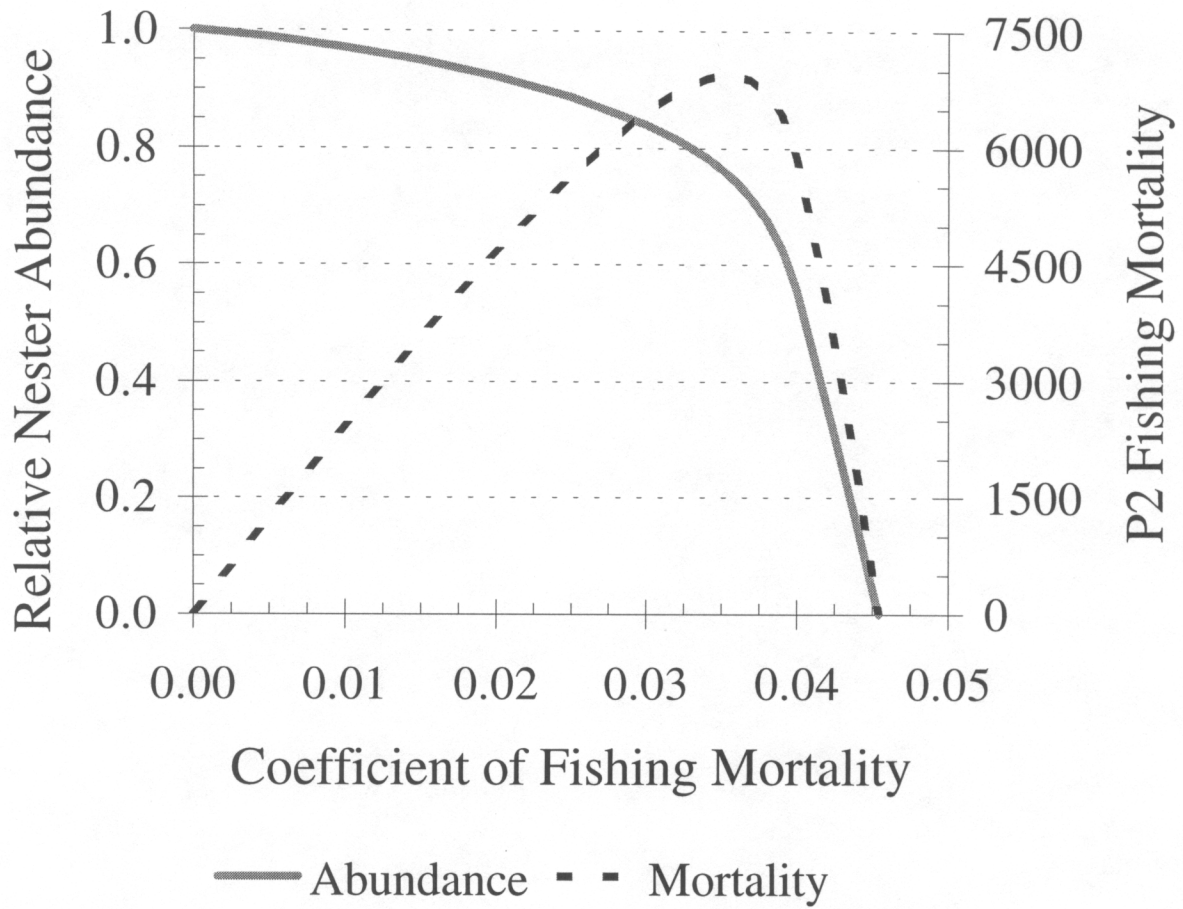


Figure 4. Relationships between the coefficient of incidental P2 fishing mortality and equilibrium levels of the P2 fishing mortality and relative nester abundance for Japanese loggerheads. Indicated equilibrium levels of P2 mortality assume a maximum nesting population of 10,000.

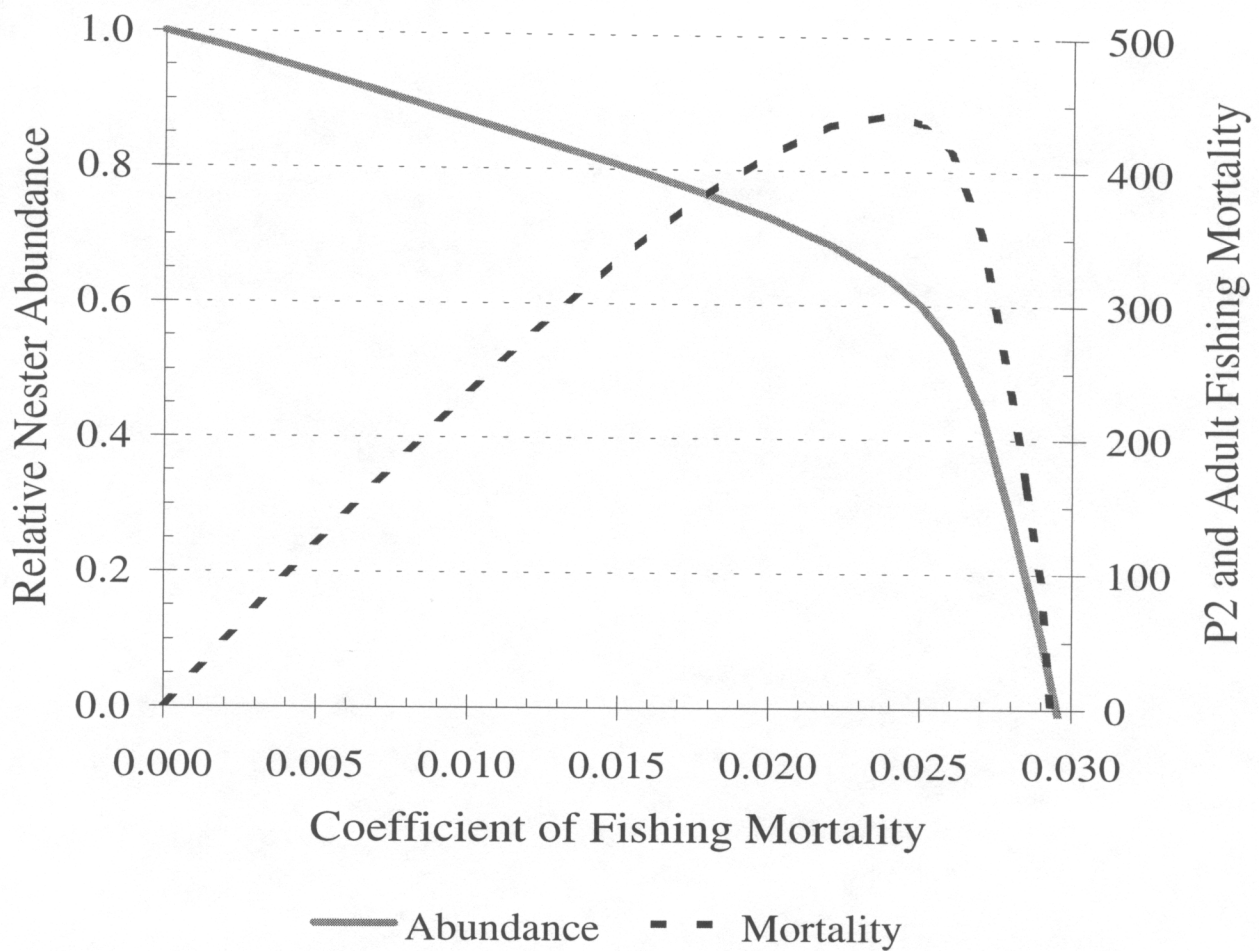


Figure 5. Relationships between the coefficient of incidental P2 and adult fishing mortality and equilibrium levels of the fishing mortality and nester abundance for Malaysian leatherbacks. Indicated levels of fishing mortality assume a maximum nesting population of 2,000 turtles. Assumes $\beta = 10.0$ and no human-caused mortality of eggs or nesters.

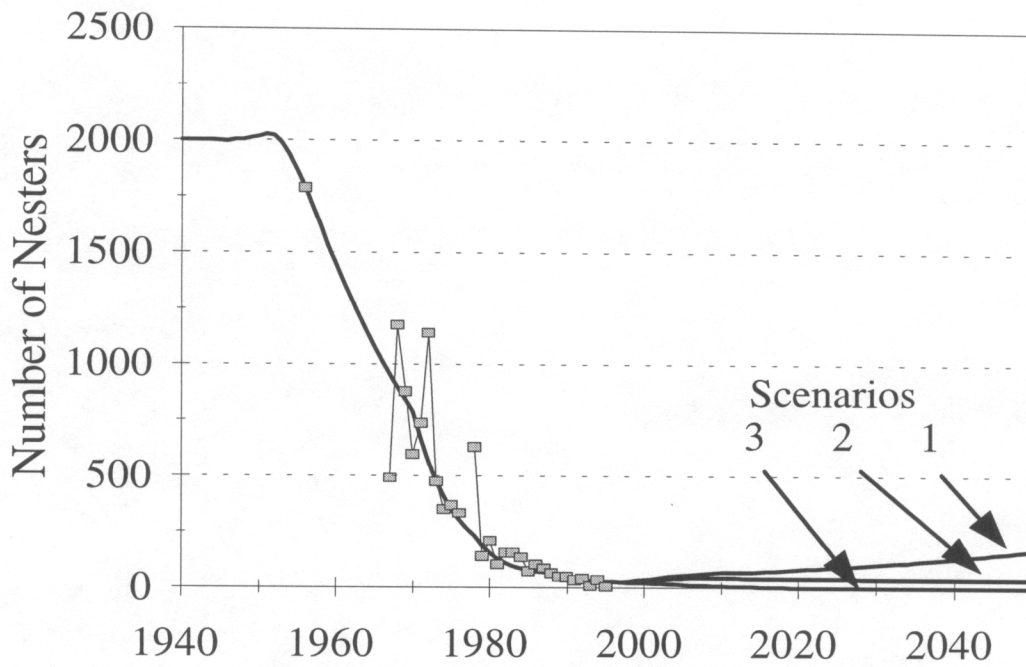


Figure 6. Projected abundance of Malaysian leatherback nesters under 3 scenarios after 1997: (1) no Hawaii longline mortality after 1997; (2) Hawaii longline mortality continues at 1997 instantaneous rate; (3) Hawaii longline mortality continues at 5 times the 1997 rate. Assumes no other human impacts occur and that all leatherbacks taken in Hawaii fishery are Malaysian.

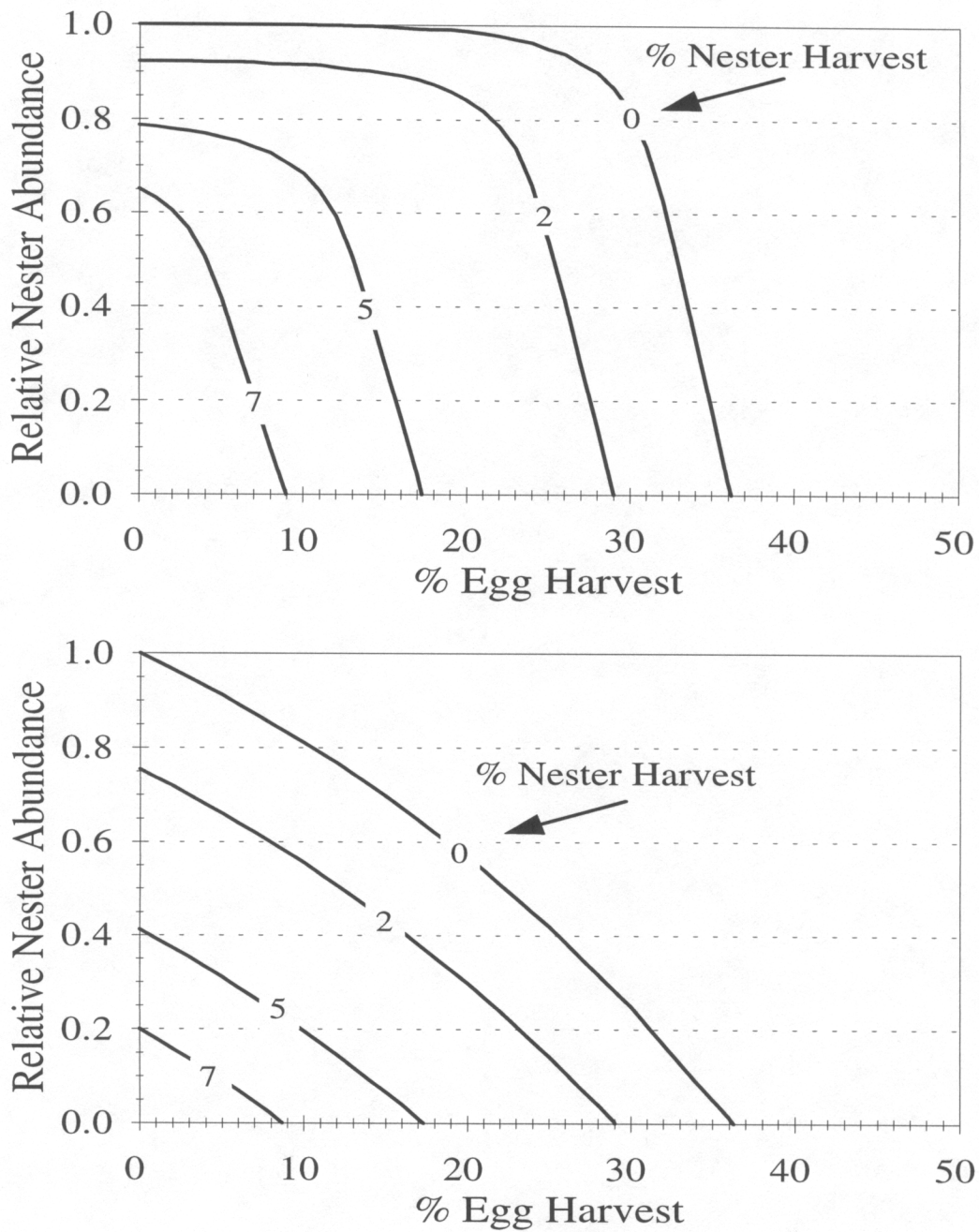


Figure 7. Equilibrium relationship between relative nester abundance and annual percentage harvest levels of eggs and nesters for Mexican leatherbacks. Top: assumes density-dependence shape parameter $\beta = 10.0$; Bottom: assumes $\beta = 1.0$.

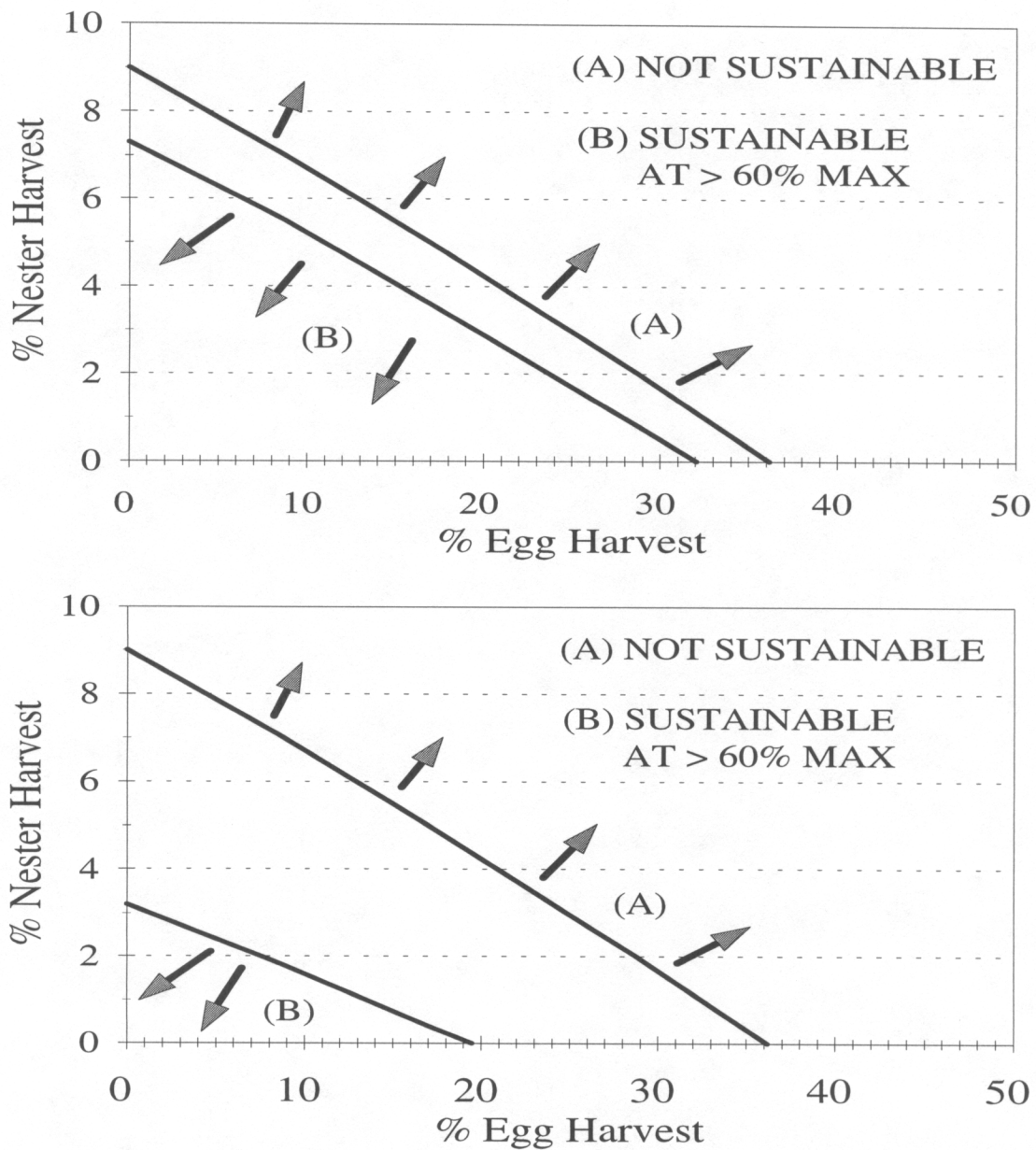


Figure 8. Region of sustainable egg and nester mortality levels for Mexican leatherbacks consistent with recovery to 60% of maximum nester abundance. Top: assumes density-dependence shape parameter $\beta = 10.0$; Bottom: assumes $\beta = 1.0$.

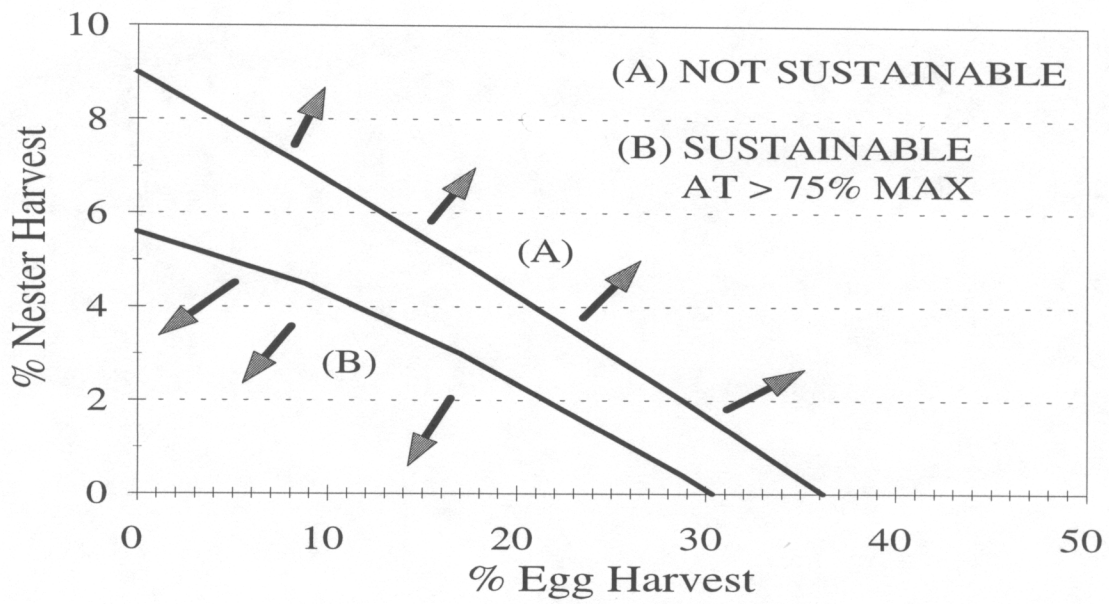


Figure 9. Region of sustainable egg and nester mortality levels for Mexican leatherbacks consistent with recovery to 75% of maximum nester abundance. Assumes density-dependence shape parameter $\beta = 10.0$.

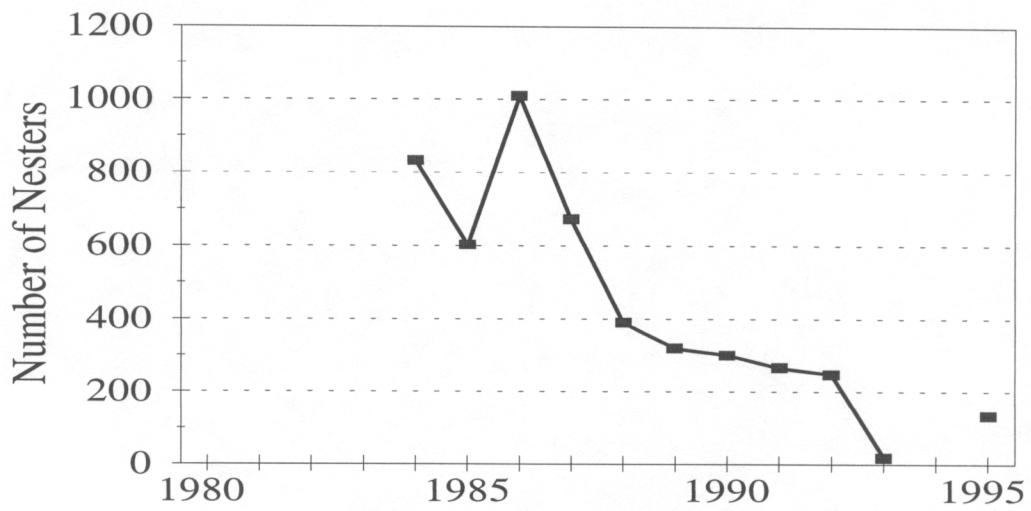


Figure 10. Estimated annual abundance of leatherback nesters at Playa Mexiquillo, Michoacan, Mexico. Derived by dividing annual nest counts reported in Sarti et al. (1996b) by an assumed constant average nest frequency.

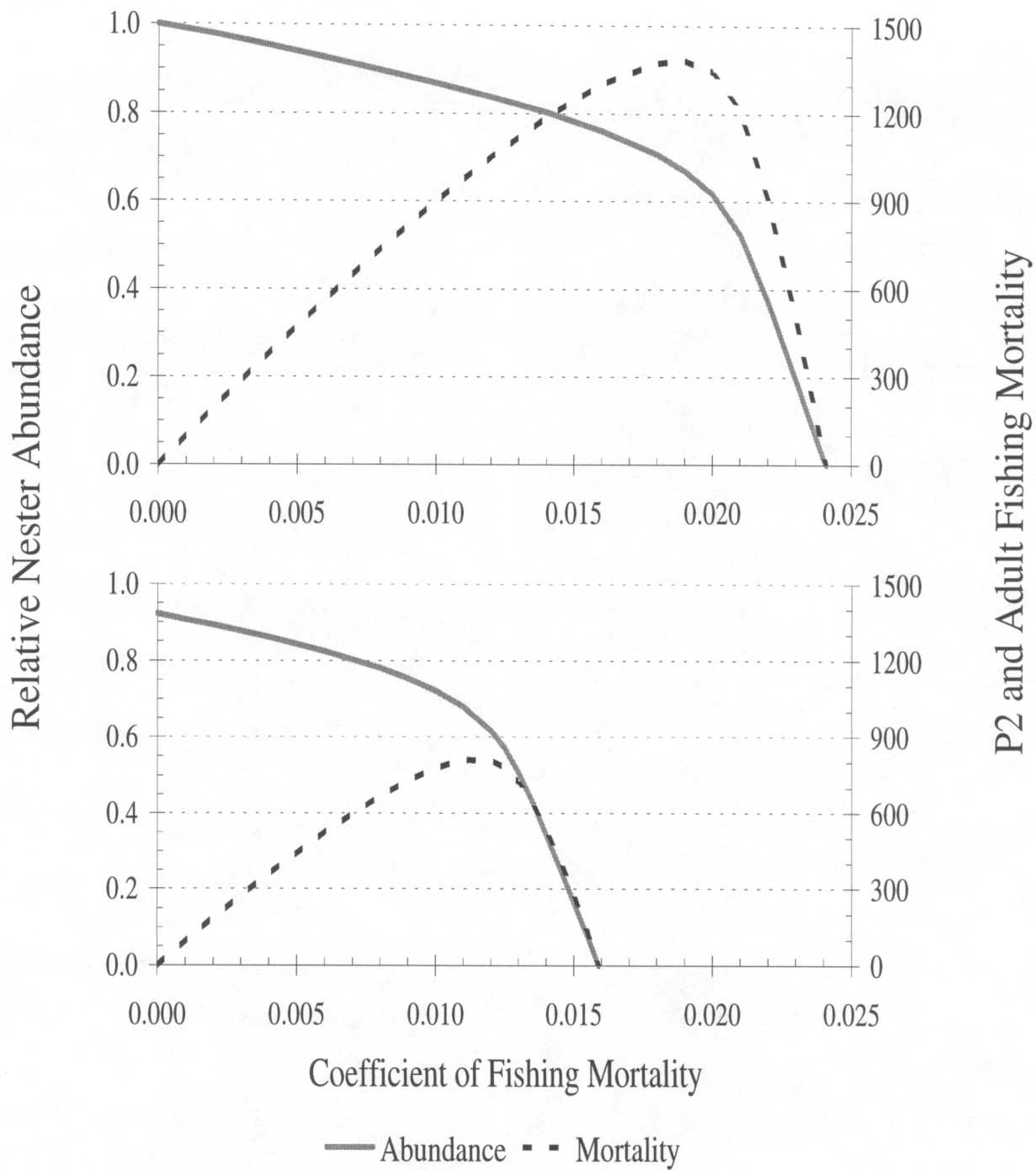


Figure 11. Relationships between the coefficient of incidental P2 and adult fishing mortality and equilibrium levels of the fishing mortality and nester abundance for Mexican leatherbacks. Indicated levels of fishing mortality assume a maximum nesting population of 10,000 turtles. Assumes $\beta = 10.0$. Top: no human-caused mortality of eggs or nesters. Bottom: 5% egg harvest and 2% nester kill annually.

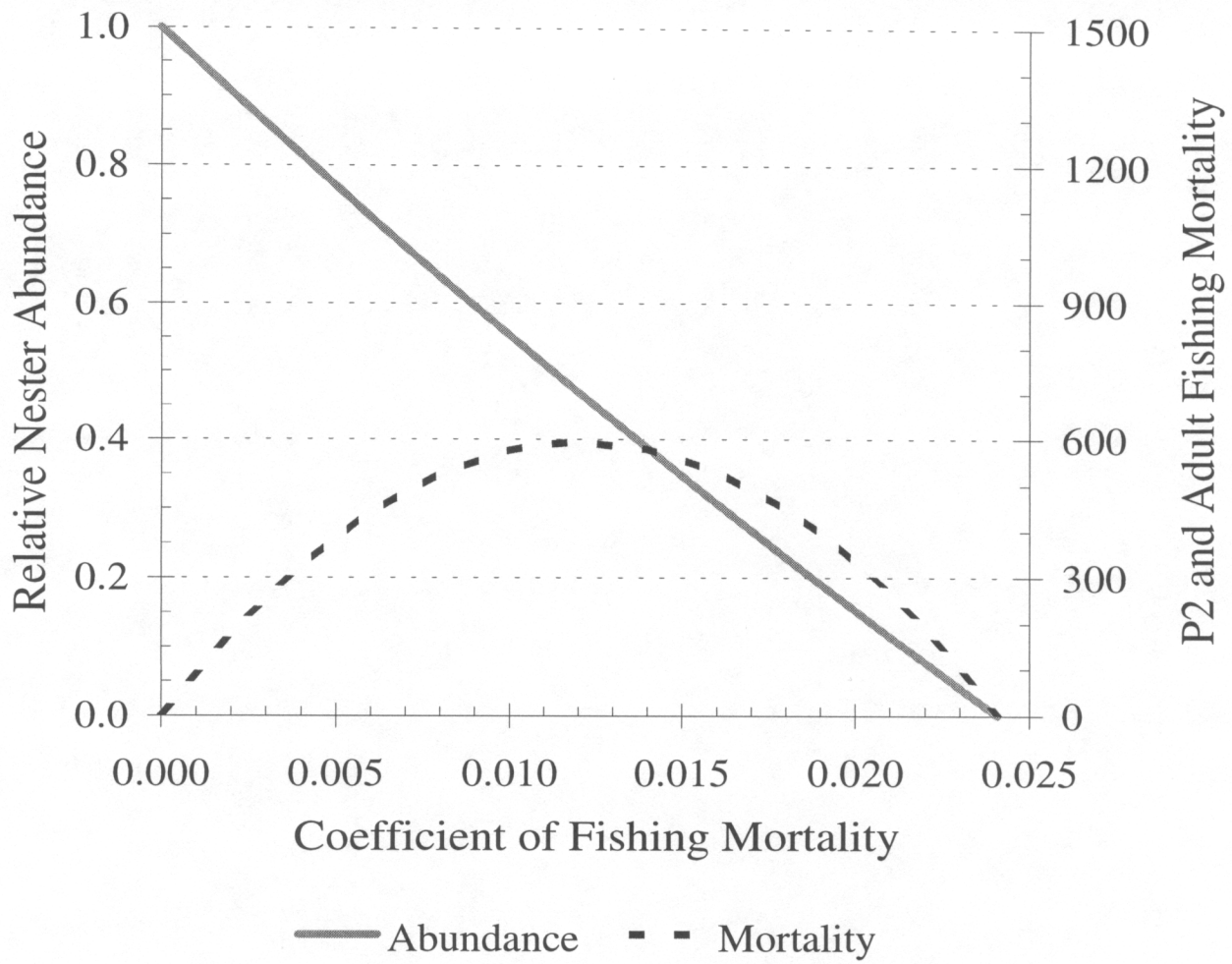


Figure 12. Relationships between the coefficient of incidental P2 and adult fishing mortality and equilibrium levels of the fishing mortality and nester abundance for Mexican leatherbacks. Indicated levels of fishing mortality assume a maximum nesting population of 10,000 turtles. Assumes $\beta = 1.0$ and no human-caused mortality of eggs or nesters.

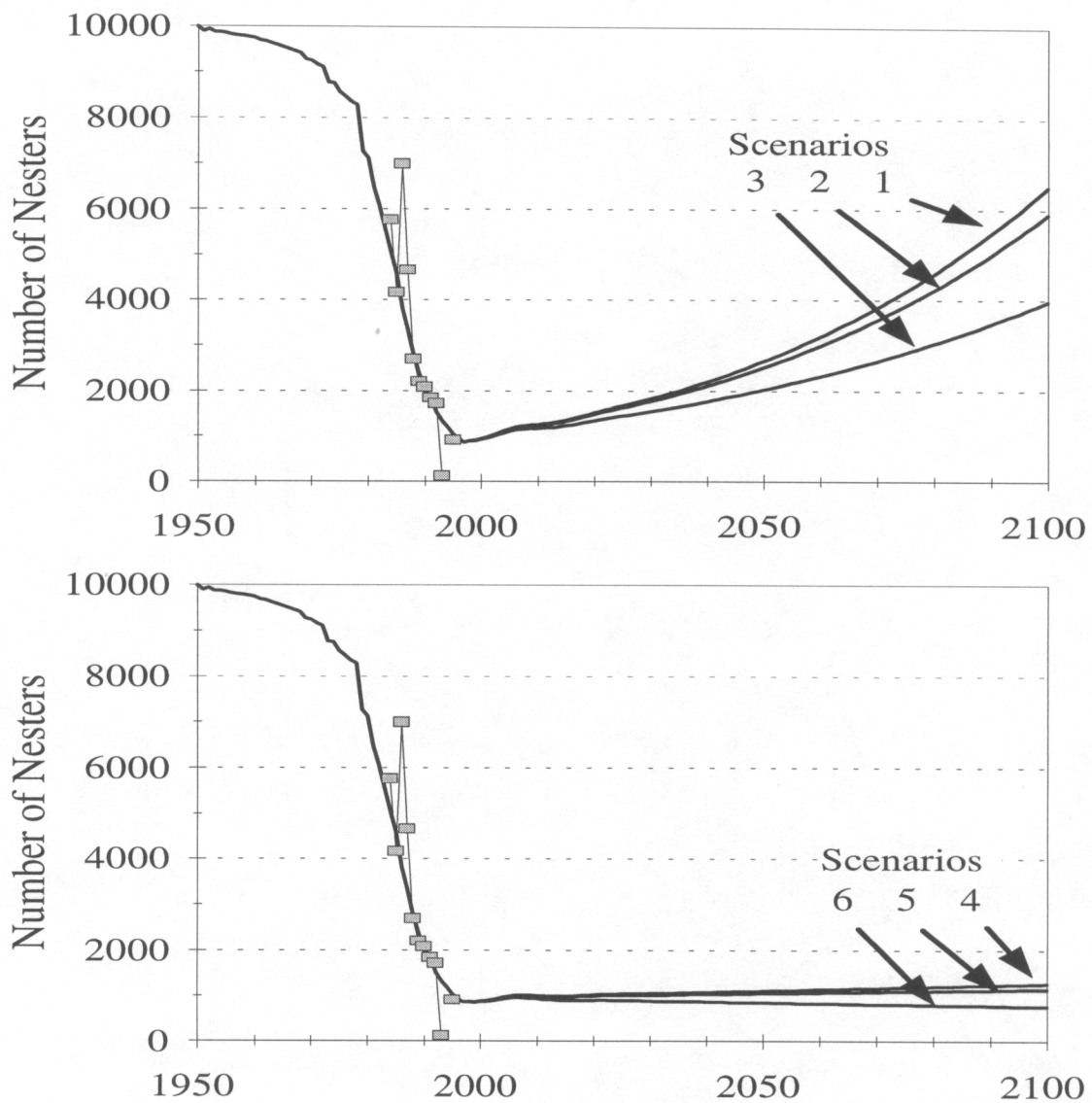


Figure 13. Projected abundance of Mexican leatherback nesters under various scenarios of human-induced mortality after 1997.

Scenario	Source of Mortality			
	Hawaii LL Fishery	Other Fisheries	Nesters	Eggs
1	None	None	None	None
2	At 1997 rate (x)	None	None	None
3	5x	None	None	None
4	None	10x	2%	5%
5	x	10x	2%	5%
6	5x	10x	2%	5%

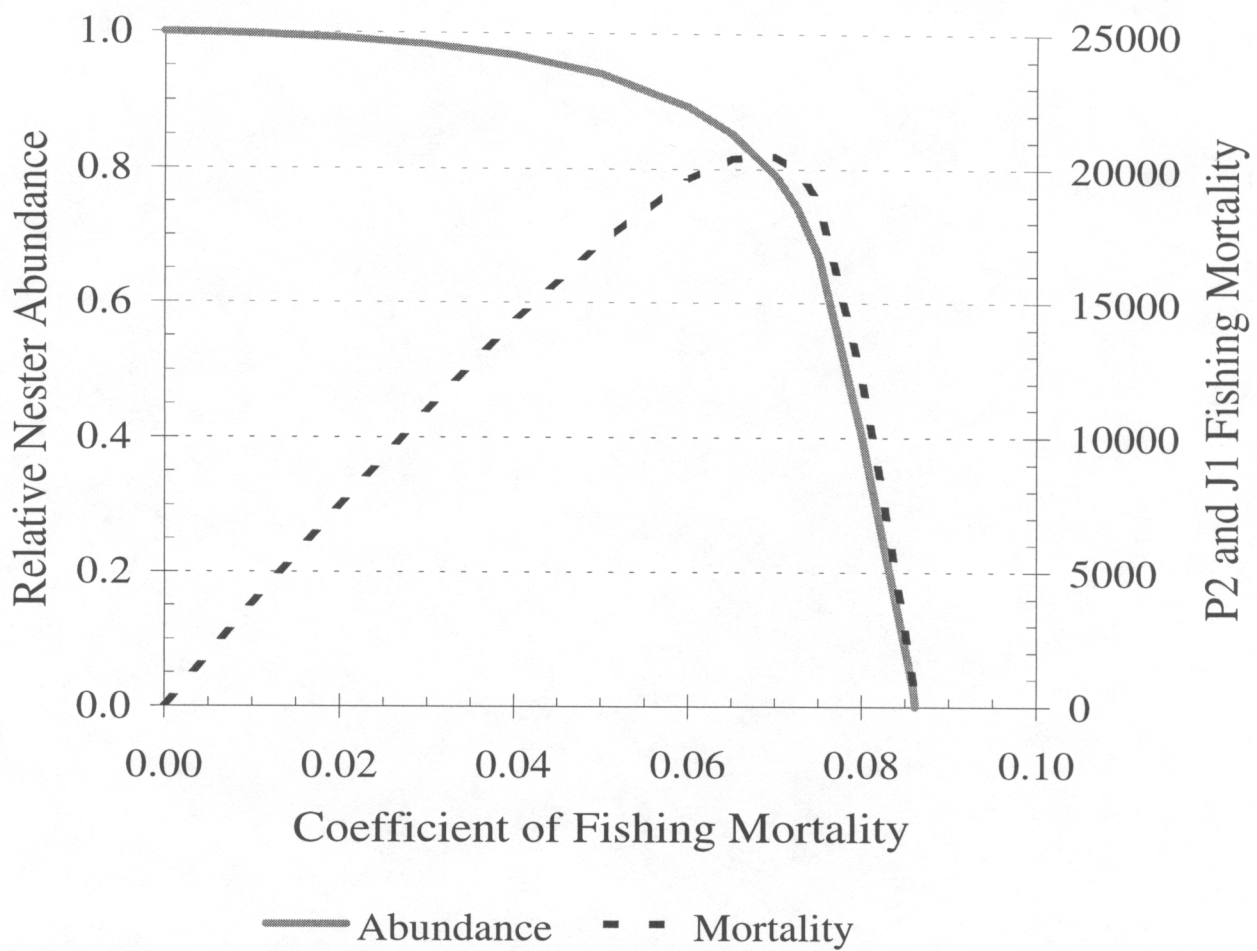


Figure 14. Relationships between the coefficient of incidental P2 and J1 fishing mortality and equilibrium levels of the fishing mortality and nester abundance for Hawaiian green turtles. Indicated levels of fishing mortality assume a maximum nesting population of 5,000 turtles. Assumes $\beta = 10.0$ and no human-caused mortality of eggs or nesters.

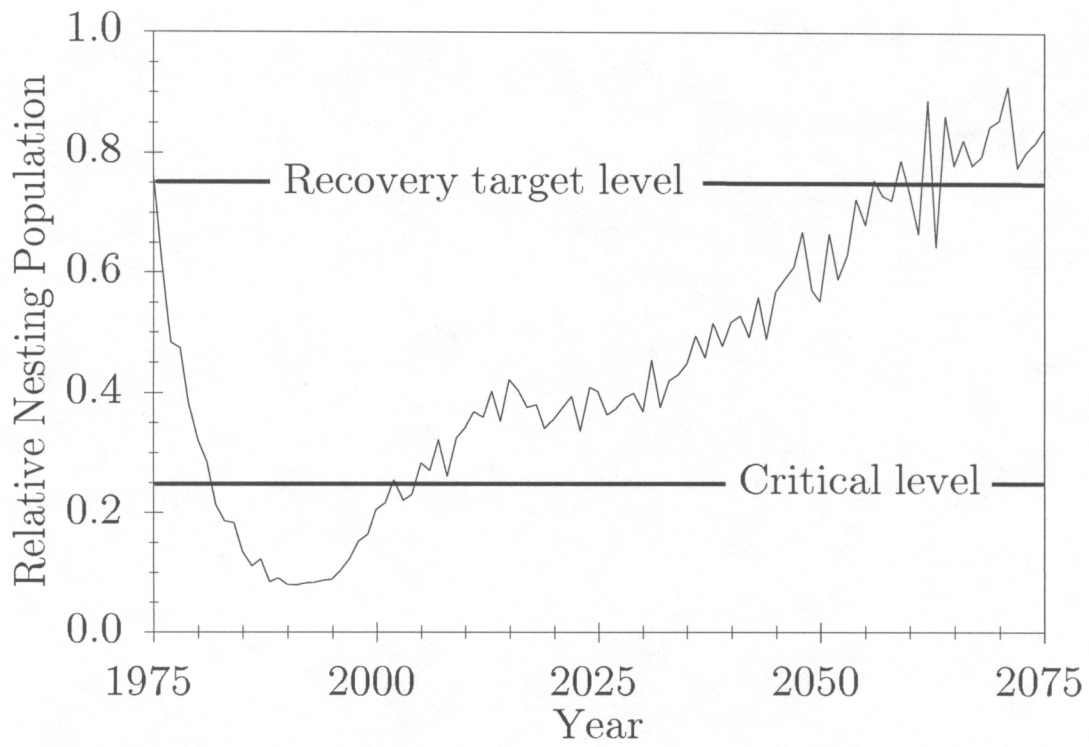


Figure 15. Hypothetical trajectory of a recovering nesting population and strategic monitoring thresholds. Recovery target level is 75% of maximum ("pristine") nester abundance.